Soil Erosion in Europe

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Soil Erosion in Europe

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To Brenda and Cati for their constant support and to the late Jan de Ploey for his fundamental influence on soil erosion studies in Europe

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Preface

This book has grown directly from a network of European researchers set up under the aegis of COST (Co-operation in Science and Technology), largely funded by the European Union, and running from 1998 to 2003. Funding for the COST Action allowed researchers from 20 countries to meet three to four times a year in workshops, conferences and small groups to discuss issues of soil erosion around the broad theme of Soil Erosion and Global Change (COST 623). Many of the 114 contributors to this book were partners in the COST Action.

The book also grew from reflections and comments made by several experts [Jan de Ploey, R.P.C. Morgan, Mr Denis Peter (EU DG XII)] about the need for an overview of the extent, seriousness and impact of erosion in Europe. It comes at a time when Europe is, for the first time, developing a coherent soil protection policy. Another important political development, not unrelated to erosion, is the reform of EU agricultural policy driven by overproduction, excessive expense and concerns about environmental degradation and contamination. Reform of the Common Agricultural Policy and the new Agri-Environment measures has put the emphasis on the control of soil erosion and sediment pollution and the management of European landscapes in a more sustainable manner.

No comprehensive assessment of processes, rates, spatial distribution and significance of soil erosion exists for Europe. The literature is scattered and sometimes superficial. This book is unique in that it presents soil erosion assessments largely based on field observations and measurements throughout Europe, rather than on estimates using erosion models. The review considers on-site and off-site effects and erosional hotspots. The book aims to be of value to researchers, high-school teachers, students, policy-makers and all those involved in environmental protection.

The book consists of two parts: (1) an overview of soil erosion processes and problems in each country and (2) cross-cutting themes. The major erosion processes affecting arable land and noncultivated land are covered: water erosion, wind erosion, shallow landsliding, tillage erosion, soil losses due to root and tuber harvesting, land levelling, piping and physical degradation (surface sealing, crusting and soil compaction), major erosion factors, impacts, erosion models and government and agency response.

There are two important qualifications or explanations. First, in some countries the amount of data is minimal either because the subject of soil erosion has not been investigated or because erosion is deemed to be of minor significance. There are therefore many gaps in our knowledge which are revealed by this survey; it will be instructive to repeat the review perhaps in 10 or 15 years time.

Second, discussion of soil protection measures is limited for several reasons. It was felt that (a) a survey of erosional processes and their areal extent was already important in itself and therefore sufficient for one volume and (b) that soil conservation was much less investigated, and that this would be a more appropriate subject for review by members of COST 634 (On and Off-site Environmental Impacts of Runoff and Erosion: 2004–08).

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Section 1

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1.1

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1.1.1 INTRODUCTION

Norway is situated between 58 and 71 ° N and between 5 and 31 ° E. A north–south mountain range, with an elevation up to 2469 m, divides the country into a steep western side and a more gentle eastern side. The Gulf Stream has a meliorating impact on the climate. Yearly precipitation ranges from 278 to 3575 mm and average temperature ranges from +7.7 °C (south-west) to –3.1 °C (Finnmarksvidda in the north).

During several glacial periods Norway was covered with glaciers. After the ice disappeared, the south eastern part of the country was covered by sea. The most important deposits in Norway are bare rock, marine sediments, till, fluvial and glacial river deposits. The marine deposits are dominated by clay and silt and these are also the areas with highest erosion risk. The dominating soil types reflect the acid origin of the soil. Apart from Leptosols, the dominant soil types are Podzols.

Mountains and lakes cover 75% of the country, productive forests 22% and farmland 3%, whereas built-up areas cover less than 1%. The most important agricultural crops are grass, cereals, oil seed and potatoes. Fruit, berries and vegetables are produced locally if climate and soil conditions allow. Cereals and oil seed constitute 38% of total cultivated land, cultivated grassland 56%, potatoes 1.7% and root crops and green fodder 2.2%.

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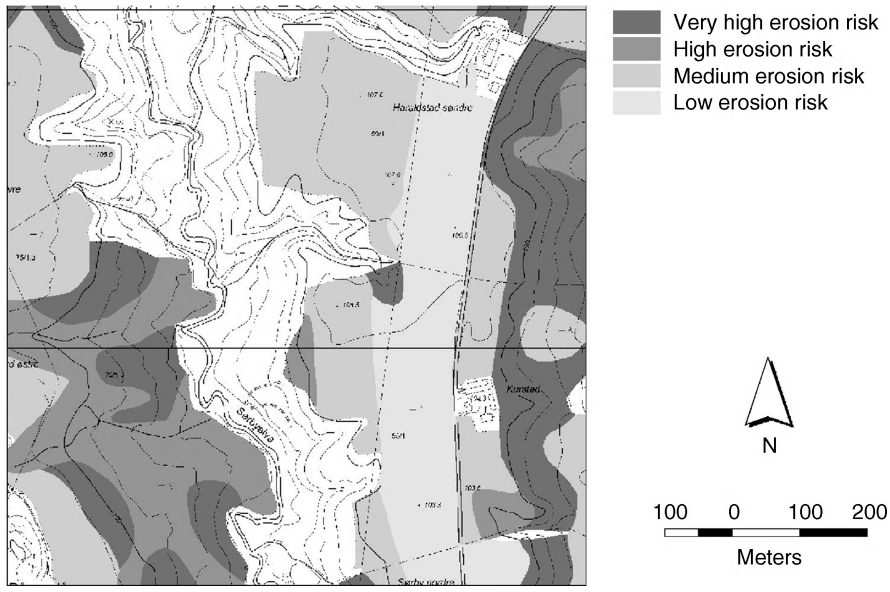

Figure 1.1.1 Map of potential erosion risk with autumn ploughing; example from Våler, Vestfold county. Low risk ($<0.5\,t\,ha^{-1}$), medium risk ($0.5–2\,t\,ha^{-1}$), high risk ($2–8\,t\,ha^{-1}$), very high risk ($>8\,t\,ha^{-1}$)

1.1.1.1 Soil Mapping in Norway

In 1988–89 an algae disaster caused the death of many sea animals in the North Sea and Skagerak. The pollution of water by nitrogen and phosphorus was indicated as the cause of the explosion of poisonous algae. The European countries bordering the seas agreed upon a 50% reduction of this pollution (North Sea Declaration) from 1985 to 1995. In Norway a reduction of erosion (P source) was politically prioritized and a soil-mapping programme was initiated for the watersheds feeding into North Sea and Skagerak.

The USLE (Universal Soil Loss Equation) model was adapted to Norwegian conditions. Erosion risk maps are produced based on soil and slope characteristics (from the soil mapping programme) and the USLE equation (Hole, 1988; Lundekvam, 1990; Arnoldussen, 1999). Figure 1.1.1 shows an erosion risk map from Våler, county Vestfold.

Four erosion risk classes are distinguished on the erosion risk maps. Of the soil mapped, 22% falls in the low erosion risk class ($<0.5t\,ha^{-1}$), 54% in the medium-risk class ($0.5–2\,t\,ha^{-1}$), 18% in the high-risk class ($2–8\,t\,ha^{-1}$) and 6% in the very high erosion risk class ($>8\,t\,ha^{-1}$). Today farmers receive subsidies when they, e.g, reduce tillage. The level of subsidy is related to the erosion risk class of the land. The soil erosion risk maps are used directly by farmers, advisory services and authorities for planning of soil erosion measures and giving subsidies.

The soil mapping activity has been concentrated in the grain production areas in the southern and south-eastern parts of the country and in the Trondheimsfjord area in mid-Norway. These areas with cereal production and marine sediments are most prone to erosion. Today, an approximately 4700 km^2 agricultural area has been mapped, which is about 50% of the total agricultural area in Norway. However, most of the area which drains to the North Sea is mapped.

1.1.2 HISTORICAL EVIDENCE OF EROSION

Historically, the marine areas had a higher level of erosion and some lakes were filled with sediment. A good example is the delta of Lake Øyeren, near Oslo, which was formed over many centuries. It is the result of natural erosion processes starting after the Ice Age for areas below the marine limit. However, human-induced erosion has increased considerably in modern times and both on- and off-farm consequences became clear. Sediment cores taken from Lake Øyeren document increased erosion in this area due to land use changes in agriculture. Production systems have changed from grassland and husbandry to cereal production and soil tillage in autumn. The change in production systems, which was a result of political decisions and promoted by subsidies, also resulted in intensive land levelling and caused higher erosion rates.

1.1.3 CURRENT EROSION PROCESSES

Soil erosion in Norway mainly occurs in autumn and spring. In autumn, heavy rainfall on a nearly saturated soil can cause soil loss through surface runoff. In spring, erosion is caused by heavy snowmelt, sometimes in combination with a frozen (sub) soil (Njøs and Hove, 1986; Lundekvam and Skøien, 1998; Øygarden, 2000; Lundekvam, 2002).

Both water and wind erosion occur in Norway, but it is generally believed that water erosion is the most important. Water erosion is also a problem related to the pollution and eutrofication of rivers and lakes. Wind erosion may occur owing to strong wind on dry, uncovered, sandy soils. As an example, this often happens at Jæren (south-west Norway) along the coastline where sand dunes are formed. Only water erosion has been measured in Norway and will be dealt with in the following.

Soil erosion by water in agricultural areas in Norway can be divided into the following:

A. sheet and rill erosion occurring over most of the agricultural area;
B. deeper rilling due to concentrated flow by surface runoff, which, in severe cases turns into
C. gully erosion;
D. erosion in connection with tile drains, main outlet pipes and inlet tanks to such pipes if errors have been made regarding dimensions or construction of the systems, or the systems have been damaged later.

In addition we also find the following erosion types:

E. Erosion in streams and rivers, occuring due to scouring of the bottom and banks, earth slides into rivers and soil creep narrowing watercourses;
F. erosion in glaciated areas (constituting about 1% of Norway).

Farming practices directly influence the occurrence of erosion types A–D. Erosion type E may also be affected by farmers' choices due to actions that may stabilize or destabilize river channels.

The importance of all these types of erosion differs according to natural factors such as climate, topography, soil type and vegetation, and also various human actions including agricultural activities. Sheet and rill erosion have been measured in plot experiments (Table 1.1.1) over many years (Njøs and Hove, 1986; Lundekvam and Skøien, 1998) and in small agricultural catchments (Lundekvam, 1997; Øygarden, 2000) on different soil types and under different cultivation systems. This research (locations are given in Figure 1.1.2), show that surface runoff and erosion risk on agricultural areas in south-east Norway generally were highest during late autumn, winter and spring owing to surface runoff because of frost in the soil and/or saturated soil. This seasonal distribution of soil erosion risk over the year, which affects all types of erosion, implies that no-till

TABLE 1.1.1 Sheet and rill erosion measured on plots at five sites in south-east Norway, 1992–2000 (Lundekvam, 2002). Precipitation was 7% higher and temperature 0.9 °C higher than the 1961–90 average. Soil types: I, levelled silty clay loam with low content of organic matter (OM); II, clay soil with higher OM; III, loam with high OM and high aggregate stability. Land use: Pl, ploughing; Ha, harrowing, Pl-spring, no till autumn; Di, direct drilling; Wi-wh, winter wheat after ploughing and harrowing autumn

Site location	Precipitation (mm), Temperature(°C)	Length (m), slope (%)	Soil type	Land use	Surface runoff (mm)	Soil loss (t ha^{-1})
Askim	858, 5.5	25, 13	I	Pl–autumn	263	4.36
Askim	858, 5.5	25, 13	I	Ha–spring	231	0.49
Ås	842, 6.2	21, 13	I	Pl–autumn	302	6.36
Ås	842, 6.2	21, 13	I	Wi–wheat	317	7.63
Ås	842, 6.2	21, 13	I	Ha–autumn	267	3.00
Ås	842, 6.2	21, 13	I	Pl–spring	231	0.71
Skedsmo	848, 5.8	30, 13	I	Pl–autumn	172	2.71
Skedsmo	848, 5.8	30, 13	I	Ha–spring	170	0.38
Skedsmo	848, 5.8	30, 13	I	Meadow	170	0.13
Sarpsborg	867, 7.2	22, 12	II	Pl–autumn	123	1.04
Sarpsborg	867, 7.2	22, 12	II	Wi–wheat	123	0.80
Sarpsborg	867, 7.2	22, 12	II	Ha–autumn	123	0.62
Sarpsborg	867, 7.2	22, 12	II	Di–spring	134	0.18
Ås	842, 6.2	28, 13	III	Pl–autumn	83	0.60
Ås	842, 6.2	28, 13	III	Pl–spring	153	0.11

will decrease soil losses compared with tillage in autumn. Actions against this type of erosion are thus based on solid scientific evidence. This was also the basis for governmental support for no autumn tillage.

There are no measurements of soil erosion covering all of Norway and it is not possible to quantify all the different erosion processes. However, there is no doubt that in agricultural areas processes A–D above will all be important, and these processes have been greatly increased by land levelling.

Field-scale (0.35–3.2 ha) measurements of erosion during a 6-year period in the Akershus county (Table 1.1.2) showed great variations in soil losses. For the smallest fields erosion was only measured in winters with frozen soils. The highest losses occurred after a combination of rainfall and snowmelt on partly frozen soil.

In the National Agricultural Environmental Monitoring Programme (JOVA), soil erosion and losses of nutrients and pesticides are monitored in agricultural catchments. Soil losses have been measured at the outlet of agricultural catchment areas of some square kilometres in the JOVA Programme and reported annually (e.g. Bechmann *et al*., 1999, 2001; Vandsemb *et al*., 2002). These measurements include all erosion processes (Table 1.1.3). The catchments Grimestad and Hotran have considerable erosion in stream channels. The catchments Skuterud, Mørdre, Kolstad, Grimestad and Volbu are all situated in the eastern part of southern Norway, Vasshaglona at the southern coast, Hotran in mid-Norway and Naurstad in northern Norway.

These catchments include different management systems, crops and tillage and should be representative of production systems in different regions. The catchments Skuterud and Mørdre represent areas with marine sediments and cereal production, assumed to be high-risk erosion areas.

By use of the ERONOR model (Lundekvam, 2002), the climatic erosion risk for sheet and rill erosion has been estimated in four regions in Norway where relative values compared with Aas (south-east Norway) were Aas 1, Mjøsa region 0.25, Jæren (south-west Norway) 1.9 and mid-Norway 0.77. However, owing to differences in soil types and agricultural practices, the resulting erosion rates in these areas including erosion

Figure 1.1.2 Locations of erosion measurements in Norway

types A, D and roughly type B using the ERONOR model and the EcEcMod modelling system (Vatn *et al*., 2002) were estimated to be Aas 0.94, Mjøsa region 0.19, mid-Norway 0.54 and, Jæren 0.11 t ha^{-1} yr^{-1} for the period 1976–97.

Soil losses through tile drains have been measured by Lundekvam (1997) and Øygarden *et al.* (1997). Tillage practices, soil type, conditions at the time of drainage, drainage equipment and time since drainage are the main factors that affect these losses. Øygarden *et al*. (1997) have shown how surface water and soil

TABLE 1.1.2 Surface runoff (mm) and soil loss (t ha^{-1}) measured at field scale in Ullensaker community, Akershus county, in the period 1987–92 (Øygarden 2000). Mean precipitation = 825 mm

Field No.	Area (ha)	Slope length (m) Gradient (%)	Soil type	Land use	Surface runoff (mm)	Soil loss (t ha^{-1})
1	0.36	100 12	Silty clay loam	Cereals	31–172	0.07 –1.5
2	3.25	175 3–14	Silty clay loam	Cereals	29–128	0.03–1.6
3	0.41	113 12	Silty clay loam	Cereals	10–161	0.01–0.1
4	0.35	75 14	Silty clay loam	Cereals	9–77	0.08–0.1
6	0.86	155 4–9	Silty clay loam	Cereals	158–292	0.20–2.6
8	0.44	113 6–16	Silty loam	Cereals	130–327	0.2–5.2

particles can quickly find their way through macropores to tile drains on levelled soil. Measured losses have been between 0.03 and 1 t ha^{-1} yr^{-1} through drainage systems.

Erosion in deeper rills and gullies has been observed several times, but only individual studies document processes and erosion amounts. Lundekvam (1997) found in a 2.7 ha agricultural catchment on levelled soil in south-eastern Norway that erosion due to concentrated flow in valley depressions constituted 40% of total erosion in autumn ploughed fields used for grain production. Under no-till in autumn, this kind of erosion almost disappeared.

During an extreme erosion event in the winter of 1990, severe erosion with rills and gullies was widespread in the eastern Norway. In a field survey (25 fields) in three counties (Akershus, Østfold and Telemark), rills and gullies were measured (Øygarden, 2000, 2003). The combination of frozen subsoil, saturated soil with low strength, snowmelt and intense rainfall led to gully development. Gullies developed to the depth of the drains, which equals soil losses of more than 100 t ha^{-1}. Such soil losses were measured in all three counties. Locations with low clay content and high silt/sand content had highest erosion. Human activity had a significant influence on the soil losses where there was lack of surface water control. Autumn-tilled soil, winter wheat and harvested early potatoes had high erosion, whereas adjacent stubble fields had no visible erosion.

In the Skuterud and Mørdre catchments in Akershus county, a field inventory of erosion with detailed measurement of rills was carried out from 1990 to 2002 (Øygarden *et al*., 2003). Erosion patterns are dependent on management practices, topography and soil type. Rills up to 1.5 m wide and 0.70 m deep have been measured. The field inventory also documented erosion around hydrotechnical equipment, bank side and erosion in waterways, which can contribute significantly to the total soil losses at the catchment scale. This kind of erosion can be reduced by managing concentrated flow by, e.g., grassed waterways or inlet tanks for surface water combined with no-till in the bottom of the small valleys in fields where water concentrates. These and similar findings form the basis for several recommendations by advisors and subsidies given by government to reduce erosion.

In some cases, erosion in watercourses may be considerable. Bogen *et al*. (1993) investigated this type of erosion in a catchment of 659 km^2 at Romerike in south-eastern Norway. About 58% of that area is below the marine limit of 205 m. About 35% of the area was cultivated land of which 20% had been artificially levelled. Most of the smaller streams had not established a stable slope, and were scouring the bottom and banks. Frequently, bank segments slid into the creeks. Bogen *et al.* (1993) found that this natural erosion was of the same order of magnitude as erosion from agricultural land in this district. In some cases, re-establishment of damaged vegetation zones may stabilize the banks of rivers and creeks, but scouring and slides also occurred in forested areas with little or no human activity. Consequently, only limited control can be exerted on this natural process.

TABLE 1.1.3 Measured soil loss in agricultural catchments from different parts of Norway in the Agricultural Environmental Monitoring Programme, JOVA (Vandsemb *et al*., 2002)

Catchment	Community	Area (ha)	Cultivated (%)	Precipitation (mm)	Soil type	Production	Period	Soil loss ($t\,ha^{-1}yr^{-1}$)
Skuterud	Aas	449	61	785	Silty clay loam	Grain	93–02	1.7
Mørdre	Nes	680	65	665	Silt and clay	Grain	91–02	1.2
Kolstad	Ringsaker	308	68	585	Sandy loam	Grain	85–02	0.2
Grimestad	Stokke	185	43	1029	Sand	Grain/grass	93–02	3.5
Hotran	Levanger	1940	80	892	Silty loam, silty clay	Grain/grass	92–02	2.6
Naurstad	Bodø	146	35	1020	Sand, peat	Grass	94–02	0.9
Volbu	Ø. Slidre	168	41	575	Silty sand	Grass	92–02	0.1
Vasshag	Grimstad	65	62	1230	Sand	Grain/vegetables	92–02	1.4

Erosion rates will differ over time as a result of changes in land use, climate change, etc. Bogen *et al*. (1993) measured sedimentation rates on flood plains in the lower part of the River Leira catchment in south-east Norway. The rates were 2.4 cm yr^{-1} for 1954–85 and 4.3 cm yr^{-1} for 1986–90. Land levelling, more autumn ploughing and severe floods in the last period were the most obvious reasons for the increase in sedimentation rates.

Erosion rates in glacial rivers from five glaciers in the period 1967–76 amounted to 1.9–27.4 t ha^{-1} yr^{-1} (Otnes and Ræstad, 1977). In contrast, erosion rates in rivers from woodland with till soils seldom go beyond 0.06 t ha^{-1} yr^{-1} and in non-glaciated mountain areas seldom beyond 0.1 t ha^{-1} yr^{-1} (Bogen and Nordseth, 1986). However, in catchments below the marine limit with clayey and silty soils with agriculture and land levelling, erosion rates may be high. In the River Leira, Bogen *et al*. (1993) measured a rate of 2.15 t ha^{-1} yr^{-1} from the area below the marine limit for the period 1983–92.

1.1.3.1 Snowmelt Erosion

In Norway, the winter and the snowmelt period are often the most important periods for runoff and soil loss (Lundekvam and Skøien, 1998; Øygarden, 2000) (Figure 1.1.3). Different runoff conditions can occur during the winter period:

- snowmelt on unfrozen soil;
- snowmelt on frozen soils;
- rainfall and snowmelt on frozen or unfrozen soil;
- rainfall on frozen or unfrozen soil.

Figure 1.1.3 Winter period with frozen soil and snowmelt can be the most important erosion period. Ullensaker, Akershus county

During snowmelt, thawing in the daytime and freezing at night result in a diurnal runoff pattern. If snowmelt occurs on unfrozen soil, a major part of the runoff can infiltrate and give small amounts of surface runoff (Øygarden, 2000). For the smallest fields in Table 1.1.3, surface runoff did not occur in years when snowmelt occurred on unfrozen soils. Annual surface runoff in such years varied between 10 and 242 mm. Surface runoff and erosion only occurred on fields with valley depressions or on levelled soil. The winter season contributed between 47 and 100% of annual runoff for these fields.

When snowmelt occurs on frozen soils, infiltration is restricted and the amount of surface runoff increases. All the above-mentioned fields had surface runoff and erosion in the years when snowmelt occurred on frozen soil. Runoff during the winter period and snowmelt are also dependent of soil moisture conditions the previous autumn. Low saturation of the soil at the onset of the freezing period and low snowmelt rates can result in higher infiltration and a smaller amount of surface runoff. Detailed studies by Lundekvam and Skøien (1998) from plot and catchment studies in the same areas as these fields showed low surface runoff and high drainage runoff due to infiltration. They also found that for winters with frozen soils and little snow, low permeability gave high surface runoff.

Unstable winter conditions with several freezing and thawing cycles are most favourable for erosion. Frozen soil restricts infiltration and rainfall or snowmelt gives high surface runoff. The topsoil might be saturated, aggregate stability and soil shear strength are reduced and the combination of rainfall and snowmelt gives little stability and high soil losses in surface runoff. The combination of rainfall and snowmelt on frozen soil has given the highest soil losses and can also cause extreme events. The above-mentioned soil losses from gullies of more than 100 t ha^{-1} were caused by such an extreme event. During such snowmelt events there can be very high variations in runoff and soil losses on a daily basis. In a field of 3.2 ha, the combination of snowmelt and rainfall in January 1990 resulted in surface runoff of 111 mm during a 2-day event. During the first day, almost clear melt water ran off with a soil loss of 0.002 t ha^{-1}. The following day, 77 mm surface runoff resulted in a soil loss of 3 t ha^{-1} (Øygarden, 2000). This event has given the highest soil losses measured in erosion research in Norway, for plot, field and small catchment studies (Lundekvam and Skøien, 1998; Bechmann *et al*., 1999; Lundekvam, 2002; Øygarden, 2000, 2003).

In recent years, the use of winter wheat has increased as a cropping system with a highly variable effect on soil erosion. There are examples of higher erosion from fields with winter wheat cover during the winter period and snowmelt than from autumn ploughed fields. This is especially the situation if the crop has not established a proper plant cover before winter starts and if the soil is both ploughed and fine tilled before sowing of the winter wheat. The focus on tillage methods for growing winter cereals has therefore increased.

1.1.3.2 Research on Conservation Tillage in Norway

The investigation of tillage systems without ploughing started in Norway in the mid 1970s.

Long-term trials (12–30 years) have been performed with several forms of conservation tillage on representative soil types under varying climatic conditions. Results of these trials indicate that the time of ploughing (spring versus autumn) has little effect on crop yields, even on clay soils. Spring ploughing may, however, delay sowing somewhat and has given a higher annual yield variation than autumn ploughing on soil with high clay content (Njøs and Børresen, 1991).

No-till systems are generally successful on well-drained loam and clay soils under the relatively dry conditions in south-east Norway, but have proved to be more problematic under wetter conditions, especially on silty and sandy soils. Reduced tillage and direct drilling have been investigated in many field experiments. On average, these systems gave about equal yields compared with autumn ploughing (Børresen and Riley, 2003). Higher relative yields compared with autumn ploughing were obtained with reduced tillage and direct drilling in years with very dry weather in the first part of the growing season.

TABLE 1.1.4 Relative erosion risk associated with different soil tillage systems (Lundekvam, 2002). The two numbers for relative erosion risk on one row reflect soils with high erodibility (small numbers) and low erodibility (larger numbers)

Tillage system	Time of sowing	Relative erosion risk
Ploughing in autumn	Spring	1.00
Harrowing in autumn	Spring	0.50–0.70
Ploughing in spring	Spring	0.14–0.35
Harrowing in spring	Spring	0.12–0.30
Direct drilling	Spring	0.11–0.25
Ploughing	Autumn	0.70–1.20
Direct drilling	Autumn	0.20–0.50

A reduced tillage system in which unploughed soil is harrowed in spring is advantageous compared with direct drilling because it loosens the seedbed before sowing. This allows the use of simpler and cheaper seed-drills. Furthermore, weed infestation is often lower after spring harrowing than after direct drilling (Semb Tørresen, 2002). The effect of different tillage systems on soil structure has been studied in many long-term experiments. The changeover from ploughing to a no-till system is considered to be a more radical change of practice than is varying the timing of tillage operations (e.g. autumn versus spring), with respect to the effect on soil structure. Nevertheless, many of our studies show only relatively small effects of this change on soil porosity, although air-filled porosity generally declines and available water capacity increases slightly (Riley *et al*., 1994). Penetration resistance is nearly always greater in unploughed than in ploughed soil, and this may restrict root growth in some cases, for example on sandy soil. Common to all studies is that the content of organic matter in the topsoil has increased in the absence of ploughing, with accompanying increases in aggregate stability (Riley, 1983; Marti, 1984; Børresen and Njøs 1993).

Measurements of the effects of various tillage systems on soil erosion have been conducted in Norway in field experiments since 1980 (Lundekvam and Skøien, 1998) and modified by later experiments and model evaluations (Lundekvam, 2002). On the basis of these studies, the tillage systems have been ranked according to their relative erosion risk (Table 1.1.4). Ploughing in autumn was used as the reference because it has traditionally been the most common tillage practice in Norway. The studies have shown that the best way to prevent soil erosion is to avoid any tillage operation in autumn. Winter wheat has a variable effect on soil erosion, depending on the degree of crop development in autumn. Direct drilling of winter wheat normally gives a low erosion risk.

1.1.3.3 Soil Conservation and Policies to Combat Erosion and Off-site Problems

Artificial land levelling in the period 1970–85 (promoted by subsidies) led to severe erosion problems and increased water pollution. In some municipalities, up to 40% of the agricultural area was levelled. Today, land levelling is not allowed without special permission. Njøs and Hove (1984) identified the adverse effect of land levelling on soil structure and erodibility. Their findings were later confirmed by Lundekvam and Skøien (1998) and Øygarden (2000). The effect of levelling on erosion will be greatly affected by how this operation is done. There are, however, no experiments where the effect of levelling has been measured directly, but it is well known that the topsoil is disturbed and more or less mixed with subsoil and compacted, resulting in a lower content of humus, reduced aggregate stability, reduced infiltration capacity and increased erodibility. By combining measurements and model considerations, Lundekvam (2003) estimated a 3–13-fold increase in soil erosion (sheet and rill erosion) on areas that were levelled. In addition, the levelling procedure often created longer slopes and more concentrated flow. In the first years after the onset of land levelling, concentrated flow

was not properly handled resulting in very large increases in erosion with the development of rills and gullies. More detail about the levelling is given in Chapter 2.12.

Erosion research has resulted in several governmental actions involving subsidies, new regulations, information, etc. Subsidies are given for tillage practice with low erosion risk, establishment of buffer zones, catchcrops and grass-covered waterways, sedimentation ponds and repairing erosion damage on levelled land. The government has set as a priority the reduction of the area under autumn ploughing in regions susceptible to erosion. The amount of compensation is related to the erosion risk level of the respective areas. The regulation has been successful and Norway has almost achieved the reduction of phosphorus but not for nitrogen to the North Sea and Skagerak as agreed in the North Sea Declaration (Bye and Stave, 2001; Eggestad *et al*., 2001). P losses from agriculture have been reduced by 34% and N losses by 24%. From 2003, each Norwegian farmer has been obliged to have an Environmental Plan for their farm, and measurements to reduce erosion are part of it. In exposed watersheds (e.g. Morsa watershed, located in Østfold county) special regulations have been made to solve the erosion problem. They are especially focused on the need to reduce tillage during the autumn period in areas with a high potential erosion risk. Erosion risk maps are being used as a valuable tool for the location of areas where special means should be prioritized. For some areas exposed to flooding, farmers are not allowed to do any autumn ploughing. There has also been a special focus on the establishment of buffer zones with grass and with different kind of trees.

Payments for no autumn tillage were introduced in 1991 irrespective of the erosion risk. After 1993, these subsidies were targeted on areas with significant erosion risk; the highest rate is given to areas with the highest risk class. At present, about 35–40% of the area is tilled only in spring, and current support is given at rates of €50–175 $ha^{-1}\,yr^{-1}$, varying according to erosion risk, with 90% of the support being given to areas with medium to extremely high erosion risk.

Figure 1.1.4 shows the trend of total area given subsidies for reduced tillage since 1991. There was a quick response to the subsidies for no autumn tillage in the first years after introduction of the payment. The subsidies for catch crops were increased in 2000 and led immediately to an increase in area.

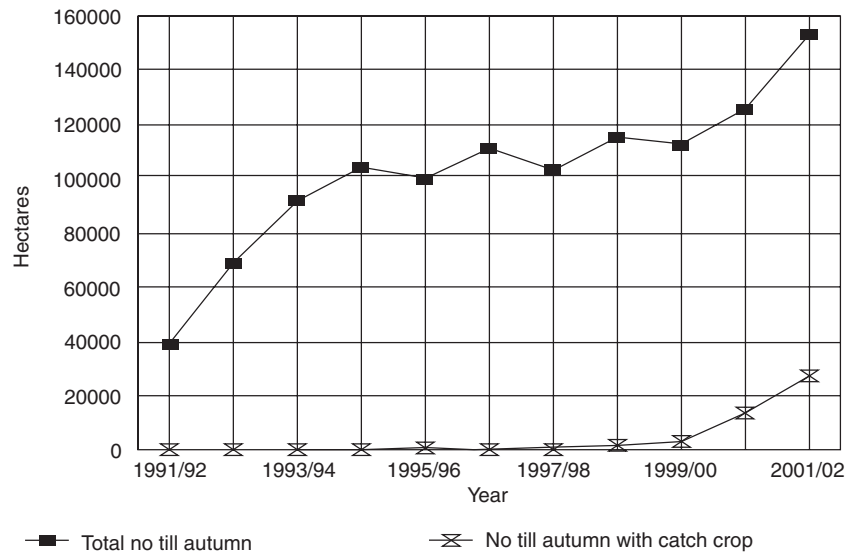

Figure 1.1.4 Total area (ha) receiving subsidies for reduced tillage (no tillage in autumn) each year and total area with no tillage and catch crops. (Reprinted from *Environmental Science and Policy*, Vol. 6, H. Lundekvam *et al*., Agricultural policies in Norway and effects on soil erosion, pp. 57–67, 2003, with permission from Elsevier)

Other environmentally motivated payments have been introduced in addition to reduced tillage. These include technical methods to control water flow to reduce erosion risk (hydrotechnical installations, grassed waterways) or capturing soil particles before reaching water bodies (buffer zones, sedimentation ponds). The number of sedimentation ponds and buffer zones being built and receiving subsidies has increased from 10 and 7 in 1994 to 88 and 15 in 2001, respectively. The farmers receive up to 70% of the cost of establishing such systems and from 1994 until 2001 the cumulative payment for these systems amounted to €591.2 and €46.2 million, respectively. Payments for repair of hydrotechnical constructions have been given from 1988. Since then, about 4500 hydrotechnical installations have been repaired (cumulative payments for this amounted up to €7.7 million). Installations which do not function may lead to intense gullying or other kinds of erosion due to concentrated flow.

At the beginning of the 1990s, research on the effects of vegetative buffer zones and sedimentation ponds was initiated in Norway. It was a normal procedure to do autumn ploughing and other kind of tillage as near to streams as possible. Trees and other vegetation near the stream banks were often removed. Because of visible erosion on agricultural land and joint efforts to implement measures to reduce erosion, a new focus was placed on retention areas in the landscape. Sedimentation ponds have shown to be effective in reducing sediment transport (Braskerud, 2001). Ponds with a size of less than 0.1% of the catchment area have reduced sediment transport by 50–60%. A major reason for the effectiveness is that particles are often transported as aggregates. The establishment of several smaller ponds along the streams has therefore proved to be more effective than larger ponds at the outlet of a larger stream.

Buffer zones have shown to be effective in encouraging deposition during winter periods and snowmelt (Syversen, 2002). Buffer zones 5–15 m wide reduced sediment transport by between 55 and 95%. During periods with high surface runoff, larger particles and aggregates are transported and they sediment more easily in the buffer zone. Because of these results, farmers can receive 70% subsidies for the establishment of buffer zones and sedimentation ponds. It is recommended to use 5–10 m wide buffer zones.

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1.2

Sweden

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1.2.1 INTRODUCTION

Sweden is situated in northern Europe between latitudes/longitudes 55–69° N and 11–24° W. The country borders the Baltic Sea, Gulf of Bothnia, Kattegeat and Skagerak and has borders of 1619 km with Norway in the west and 586 km with Finland in the north. The climate varies from subarctic in the north, where it is influenced by the Gulf Stream, to maritime and continental in the south. In the north, the winters are long, lasting 8–9 months, whereas in the south, they are short and the soil does not freeze every year. Precipitation in the north and along the Norwegian border and the south-west coast ranges from 600 to 1500 mm annually. In the east, precipitation seldom exceeds 700 mm annually.

Arable soils are mostly clayey, namely clay loam or other forms of loam. Soils with 25–40% clays are defined as medium clay soils and soils with more than 40% clays are defined as heavy clay soils. However, only limited areas have heavy clay soils (Figure 1.2.1). The soil consists of glacial and post-glacial sediments of different origin and characteristics. The dominant soil type along the coast of the northern and western coasts is fine silt (Figure 1.2.2). Heterogeneous clays dominate the eastern part of the country, but there are also lowland areas with silt clay. In the mountainous forest and valley districts of southern Sweden the soil is till derived from Archaean bedrock. In Scania and the islands the most common soil type is clay or loam, but there are also fine-textured soils (Steineck *et al*., 2001). The most common mineral in these clay soils is illite. In Scania there are also smectites, and in coastal areas of the west of the country 'quick-clays'.

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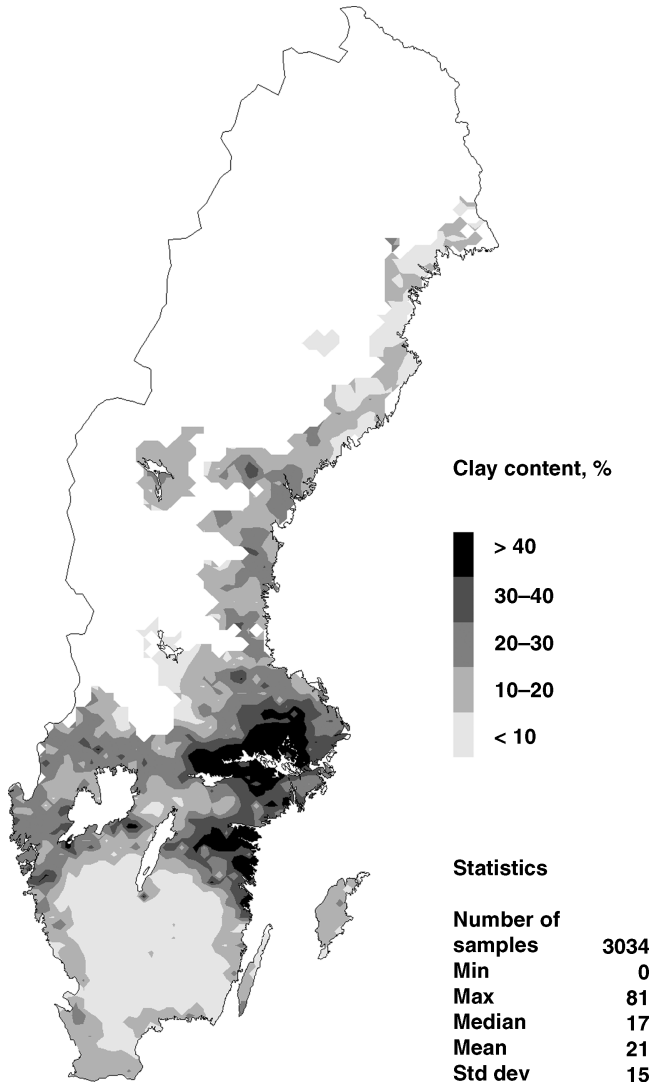

Figure 1.2.1 Clay content (%) of Swedish agricultural topsoils. (Reproduced with permission from Eriksson J, Andersson A, Andersson R. *Texture of Agricultural Topsoils in Sweden*. Report 4955, Swedish Environmental Protection Agency, Stockholm, 1999)

1.2.2 ENVIRONMENTAL CONCERN

Eroded particles are carriers of phosphorus (P) and other pollutants to surface waters and environmental concern about erosion is primarily over eutrophication. In addition, large amounts of suspended solids may cause poor light conditions in surface water that will favour *Cyanobacteria* and disturb fish breeding. A significant amount of the particles may be in colloidal form (Ulén, 2003).

The total phosphorus (TOTP) status of inland waters has recently been surveyed (Johansson and Persson, 2001). Most of the eutrophic lakes are situated in the agriculturally dominated southern and central plain areas.

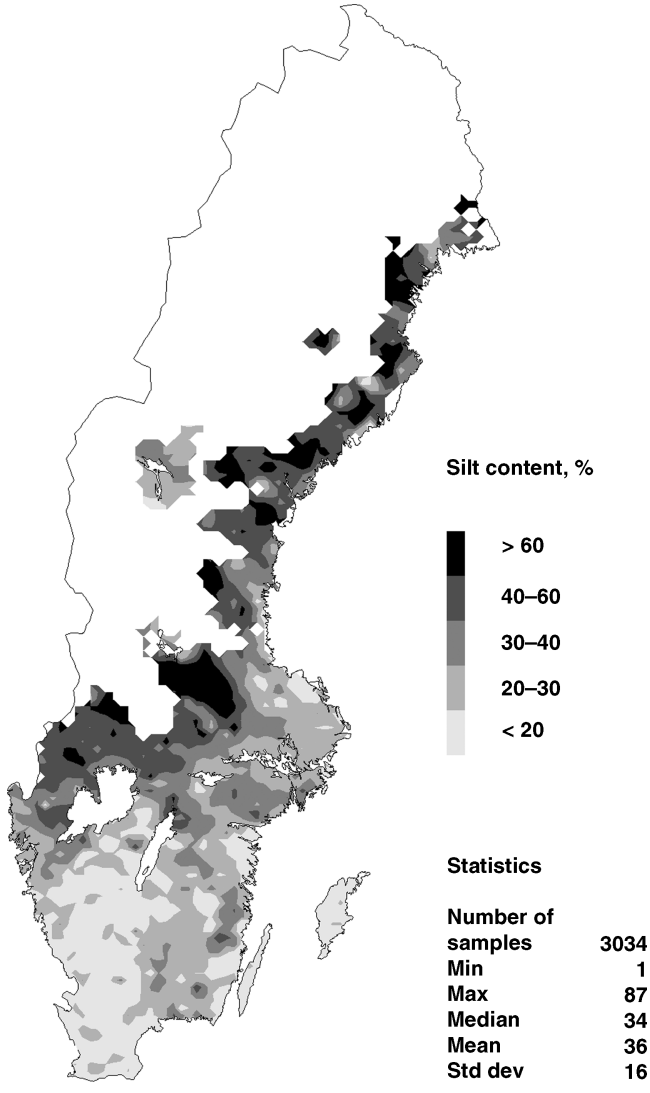

Figure 1.2.2 Silt content (%) of Swedish agricultural topsoils. (Reproduced with permission from Eriksson J, Andersson A, Andersson R. *Texture of Agricultural Topsoils in Sweden*. Report 4955, Swedish Environmental Protection Agency, Stockholm, 1999)

Small, shallow lakes have the highest P concentrations. No consensus about 'reference conditions' accounted for in the Water Framework Directive has been reached. However, in 75% of the lakes P concentrations are more than twice as high as 'comparable concentrations' based on background values as a basis for forming an environmental judgement (SEPA, 1999). The value is based on the relationship between absorbance (A_{420} nm) of the water and the total phosphorus concentration in many surface waters. It was concluded that considerable efforts are needed to reduce the P levels caused by anthropogenic activities. The average lake is shallow (<2.5 m deep) and has a low Secchi disk value (1.1 m) in addition to a high P concentration.

1.2.3 MONITORING AND FIELD MEASUREMENTS OF EROSION

1.2.3.1 Small Catchments

Until 2003 surface water in small agricultural areas was being monitored on a regular basis, but only on the basis of weekly, or twice monthly, grab sampling. Since 2003 eight areas have been selected as of special interest and are now sampled flow-proportionally. Long-term average suspended solids (SS) and TOTP concentrations in runoff from small catchments are presented in Table 1.2.1. The SS concentrations are determined after filtration through preweighed and prewashed membrane filters by weighing the filter cake. Unfortunately, different laboratories have been involved in the analysis. Membrane filters (Schleicher & Schüll, Dassel, Germany) with a pore size of 0.2 μm have usually been used but also other types of filters. One has also to keep in mind that the concentrations in Table 1.2.1 probably represent underestimated averages since they are based on infrequent sampling. Eight demonstration watersheds, all which include flow-proportional sampling of suspended solids and other parameters, are currently being used as a tool to investigate further the quality of small agricultural streams. The highest annual average SS concentration (183 mg l^{-1}) has been estimated from a stream on the east coast (county of Östergötland). High concentrations (96 and 91 mg l^{-1}) were also found in the region of Lake Mälaren and surrounding counties.

1.2.3.2 Observed Fields

The objective of the programme 'observed fields of arable land' is to monitor the influence of agriculture cultivation on the quality of surface water and groundwater within selected fields. The fields (4–32 ha) are included in the farmers' regular operations and cover various soil types, cropping and tilling regimes. The fields have measuring devices for sampling of drainage water and registration of discharge. Up to 16 experimental fields have been monitored for suspended solids (since 1986), in addition to nutrients and major constituents. The SS concentrations are all determined using membrane filters (Schleicher & Schüll) with a pore size of 0.2 μm. The concentrations are based on biweekly samples (Table 1.2.2). Generally SS concentrations in drainage water are higher than in the small streams.

Both types of waters have low concentrations of organic substances (Tables 1.2.1 and 1.2.2). Slightly higher concentrations of 'other phosphorus' (total phosphorus minus dissolved phosphate phosphorus) were indicated

TABLE 1.2.1 Number of agricultural catchments, specific area (SA) of the soil (texture), discharge, suspended solids (SS), pH, total organic carbon (TOC), total phosphorus (TOTP) and other phosphorus (total phosphorus minus phosphate phosphorus after filtration) in small streams in six regions of Sweden, flow weighted and as an average for 1977–99

Region	No.	SA[a]	Discharge (mm)	SS[b] (mg l^{-1})	pH	TOC[b] %	TOTP (mg l^{-1})	Other p (mg l^{-1})
Norrland	2	5.9	207	31	5.8	13	0.12	0.05
W Svealand, NW Götaland	6	4.2	294	44	7.2	15	0.11	0.08
Counties surrounding Lake Mälaren	7	5.9	170	41	7.6	10	0.16	0.09
South-east coast	6	3.7	139	34	7.9	9	0.19	0.07
Central Götaland	4	3.1	341	18	7.2	14	0.09	0.06
Southernmost	10	1.5	282	22	7.7	9	0.16	0.10

[a]calculated from SA = (clay fraction 8 + silt fraction 2.2 + sandy fraction 0.3) bulk density.
[b]measured during the period 1986–99.
Source: Carlsson *et al.* (2002).

TABLE 1.2.2 Number of observation fields, specific area (SA) of the soil (texture), discharge, suspended solids (SS), pH, total organic carbon (TOC), total phosphorus (TOTP) and other phosphorus (total phosphorus minus phosphate phosphorus after filtration) in tile-drained water in six regions of Sweden, flow weighted and as a long-term average 1977–99

Region	No.	SA[a]	Discharge (mm)	SS[b] ($mg\,l^{-1}$)	pH	TOC[b] %	TOTP ($mg\,l^{-1}$)	Other P ($mg\,l^{-1}$)
Norrland	2	3.6	275	12	6.2	5	0.05	0.03
W Svealand, NW Götaland	2	3.0	216	60	6.2	–	0.15	0.07
Counties surrounding Lake Mälaren	2	9.2	145	230	7.1	13	0.29	0.19
South-east coast	4	4.7	145	59	7.6	6	0.12	0.05
Central Götaland	2	3.6	230	29	7.1	8	0.10	0.06
Southernmost	4	2.7	304	58	7.3	12	0.20	0.08

[a]calculated from SA = (clay fraction 8 + silt fraction 2.2 + sandy fraction 0.3) bulk density.
[b]measured during the period 1986–99.
Source: Johansson and Ulén (2002).

in the region of Lake Mälaren and of the southernmost counties. However, low concentrations of SS were found in the southernmost streams where the soils have a low clay content.

1.2.3.3 Large Streams

In the large streams, SS is monitored on the basis of monthly values. Filter-papers (Whatman) are used for filtrations. Since these filters do not catch fine clay particles, the results cannot be compared with the results from monitoring of agricultural land.

1.2.3.4 Plots

Runoff losses of suspended solids connected to different treatments have been measured as overland flow from plots at a few sites (Table 1.2.3). They represent different time periods and different types of water collectors. Concentrations of SS were usually 10-fold those in drainage water but the concentrations of phosphorus did not differ very much. The erosion losses caused by surface runoff from experimental plots with different tilling

TABLE 1.2.3 Number of observation fields, specific area (SA) of the soil (texture), discharge, average and maximum concentrations of suspended solids (SS), total phosphorus (TOTP) and other phosphorus (total phosphorus minus phosphate phosphorus after filtration) in surface runoff from plots in different regions. All figures are flow-weighted averages but periods and number of years differ between sites

Region	No.	SA[a]	Discharge (mm)	Average SS ($mg\,l^{-1}$)	Average TOTP ($mg\,l^{-1}$)	Average Other P ($mg\,l^{-1}$)	Max. SS ($mg\,l^{-1}$)	Max. TOTP ($mg\,l^{-1}$)	Max. Other P ($mg\,l^{-1}$)
Norrlandnorth	1	4.0	195	27	0.29	0.10	52	3.26	0.59
Norrland south-west	1	4.2	78	842	0.50	0.45	1700	5.94	5.40
Lake Mälaren	1	4.9	62	350	0.49	0.22	670	1.10	0.68
Southernmost	1	4.8	6	544	0.27	0.23	805	0.89	0.62

[a]calculated from SA = (clay fraction 8 + silty fraction 2.2 + sandy fraction 0.3) bulk density.
Source: Johansson and Ulén (2002); Ulén (2003); Ulén and Kalisky (2003).

TABLE 1.2.4 Average (Ave) and standard deviation (SD) of transport (kg ha^{-1} yr^{-1}) of suspended solids (SS) and particulate phosphorus (PP) during 7 years from 22 m long plots with silty soils in the county of Dalarna

Treatment	SS (Ave)	SS (SD)	PP (Ave)	PP (SD)
Conventional autumn ploughing	644	1070	0.32	0.36
Conventional spring ploughing	223	275	0.15	0.15
No-till, except disk harrowing, autumn	365	470	0.28	0.35
Direct drilling, spring	108	98	0.14	0.10
Deep cultivation 3 times each autumn	398	576	0.24	0.23
Conventional spring ploughing and catch crops	273	361	0.15	0.17
Ley/winter wheat and autumn tilling (wintergreen[a] soil)	358	621	0.19	0.23
Extra organic material added to the soil (cut grass)	293	438	0.24	0.25

[a]Soil is not ploughed during autumn.
Source: Ulén and Kalisky (2003).

and cropping treatments have been monitored from a silty soil with 10% slope for 7 years (Table 1.2.4). The erosion differed greatly from year to year. Generally, erosion was lower when the soil was tilled during spring and not tilled during autumn. Direct drilling resulted in other problems (low yield and enhanced losses of dissolved phosphate). Increased organic concentration in the soil improved the soil structure (Ulén and Kalisky, 2003).

1.2.4 RELATIVE EXTENT OF EROSION IN SWEDEN

The situation in Sweden is that most of the clay soils are drained: 41% of all arable land is systematically drained (mostly tile-drained), 44% has permeable soils with good natural drainage and 15% may require improved drainage. The recommended depth for lateral drains is 1.0 m and the recommended drain spacing ranges from 10 to 30 m depending on hydraulic conductivity and drainage demands. Based on this, a very rough assumption indicates that a maximum of 15% of arable land is a source of surface erosion via overland flow either directly to surface waters or indirectly via surface water inlets. Such inlets are primarily installed in depressions to avoid standing water and convey the water to the subsurface drainage system.

The transport of eroded material to the watercourse is extremely difficult to estimate. In an investigation in Scania county, between 20 and 80% of eroded material from a field was estimated to leave the field. In another study, net transport out of a catchment was only 5–10% of eroded material (Mattsson *et al*., 1989). A large amount of eroded material may settle on flood plain areas close to the field (Brandt, 1982).

No trend in sediment transport was found in a special investigation from 15 representative Swedish streams between 1967 and 1994 (Brandt, 1996). It is very difficult to separate the net loss from fields and erosion from the bottom and the sides of the watercourse. In addition, the location of a field relative to the stream may be very important (Brandt, 1982).

Field measurements of water soil erosion are very few and of wind erosion even fewer. Alström and Bergman (1986) made an inventory of 29 selected fields with water erosion problems in Scania county. Various amounts of eroded material, between 0.5 and 300 t ha^{-1}, were lost from the fields. In contrast, mapping of

critical areas for erosion in south Sweden was tried using Geographic Information System (GIS) software, a slope estimator (Van Remortel *et al*., 2001) and a national digital elevation model (Andersson, 1996). The results indicated that erosion occurred only in very limited areas in the south.

In a separate study (Alström and Bergman, 1992), it was found that only 5% of the fields in Scania suffered from rill erosion but locally transport by rill erosion may be much higher than sheet erosion. In south and central Sweden, gullies were inventoried when new national reserve parks were established (Bergqvist, 1990). However, this serious form of erosion is unusual for arable land in Sweden.

High relative erosion risk areas, calculated using the USLE equation and using large-scale topographic data, indicate the erosion risk areas to be situated in the east and west part of the country (Figure 1.2.3). If most P losses are linked to erosion, these parts would also account for most TOTP losses from land. However, at the field scale no simple and direct relationship between TOTP concentration and average slope was found from the observed fields. In contrast, soil texture was related to the loss of TOTP (Ulén *et al.*, 2001). Most fields with soils >35% clay are associated with high SS concentrations (Table 1.2.1). The relationship between topography and SS concentrations is complicated and a field should be divided into different sections in order to study the erosion process and for calculation of the length of the erosion path (Djodjic and Bergström, 2005).

1.2.5 LEGISLATION AND SUBSIDIES

Legislative concern about erosion does not exist, but there is concern about phosphorus and nitrogen losses (Table 1.2.5). Locally subsidies have been given for tilling in spring and not in autumn but these have had limited success (Ulén and Kalisky, 2003).

TABLE 1.2.5 Introduction of legislation related to phosphorus losses in Swedish agriculture in recent years; 'sensitive' areas are pollution-sensitive areas in the south together with the coastal area up to central Sweden

Year	Part of Sweden	Legislation
1994	Southern half	50 or 60% of the arable land shall be 'wintergreen' (not autumn-ploughed soils, winter crops, leys, sugar beets, etc.)
1995	All	Livestock density based on phosphorus content in manure is regulated. Maximum addition of 22 kg P ha^{-1} is allowed, which is equivalent to 1.6 dairy cows or 10.5 fattening pigs
1995	Sensitive	Manure shall not be applied between 1 August and 30 November, with the exception of application before sowing of winter crops or leys
1996	Southern	Manure and other organic fertilizers shall be incorporated within 4 h of application
1996	Sensitive	In pollution-sensitive areas slurry and urine must be incorporated within 4 h of application when spreading on bare soils
1998	All	Slurry must be spread to growing crops with techniques that efficiently reduce NH_3 emissions
1999	All	Fertilizers must not be applied on water-saturated or flooded ground or on snow-covered or deeply frozen ground
1999	Sensitive	Manure application is not permitted between 1 January and 15 February
1999	Sensitive	Application of farmyard manure, with the exception of poultry manure, is allowed on bare soils, without the requirements of autumn sowing afterwards: 20 October–30 November in the counties of Blekinge, Scania and Halland, and 10 October–30 November in the coastal areas of the counties of Stockholm Södermanland, Östergötland, Kalmar, Västra Götaland and Gotland, if incorporation takes place on the same day

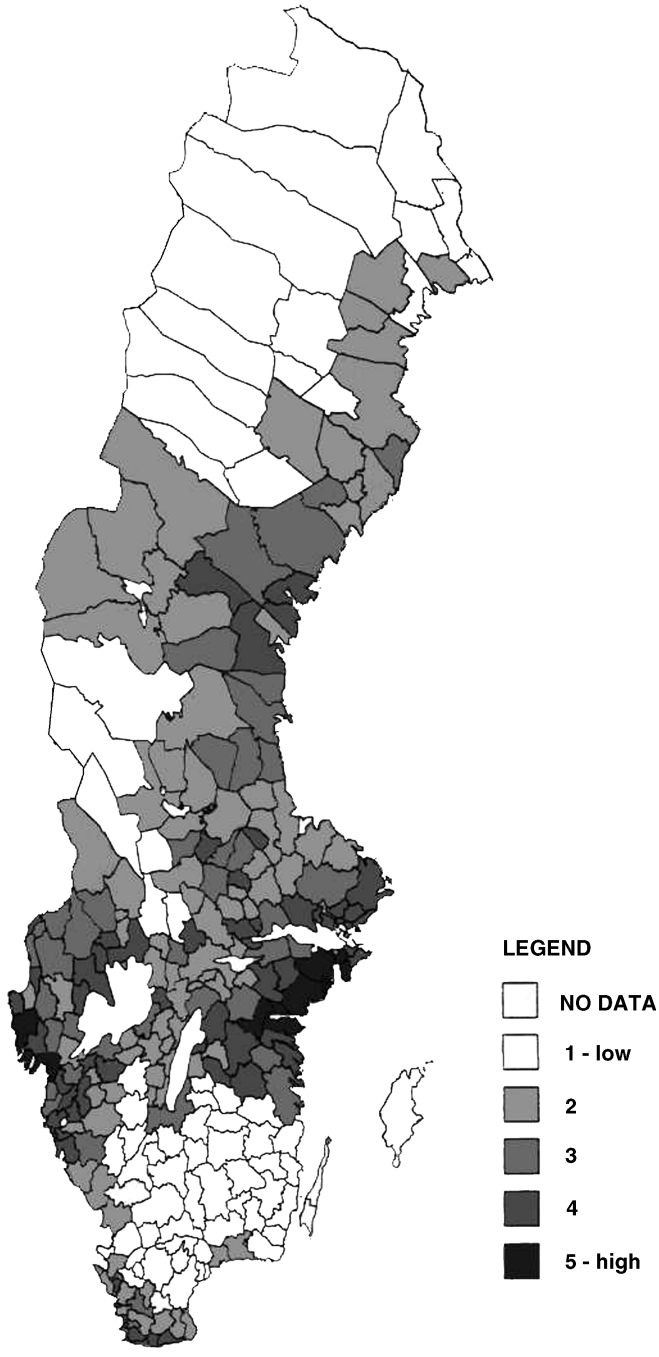

Figure 1.2.3 Relative erosion risk as a median value for municipalities weighed by the total amount of agricultural land within the municipalities (From Leek R, Rekolainen S, Tema Nord 1996: 615, reproduced by permission of the Nordic Council of Ministers)

1.2.6 SUMMARY

There have been very few studies of erosion in Sweden. Locally the problem is considerable on arable land but no group has yet done any general quantifications. Problematic agricultural areas are the heavy clay soil areas around and south of Lake Mälaren. In addition, erosion of silty soils along the coast of the northern region and the west coast might cause problems.

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1.3

Finland

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1.3.1 INTRODUCTION

Finland is the world's northernmost country producing agricultural products sufficient for its own population. It is located between the 60th and 70th parallels, and therefore differences in climatic conditions are considerable between south and north. The terrain is mostly low, flat to rolling plains interspersed with lakes and low hills. The highest point in Finland is Haltiatunturi at 1328 m. The area of Finland is 338 100 km^2, of which 27 500 km^2 (8%) is agricultural land and 68% is forest. The winters are cold; the average temperature (1961–90) in February is −5.7 °C in southern Finland and −13.6 °C in northern Finland. The average temperatures in July are 17 and 14.1 °C, respectively. The annual precipitation varies from 600 to 700 mm in southern Finland and from 450 to 550 mm in northern Finland. The length of the growing season is 165–180 days in southern Finland and 110–145 days in northern Finland.

1.3.2 GEOLOGY AND SOIL

Finnish bedrock is very old (ca 2.7–1.8 Ga), consisting mostly of acid rocks such as granite and gneiss. The bedrock is resistant and therefore weathers slowly. Soil deposits have been developed during and after the last Ice Age, hence being geologically young and thin. The average thickness of superficial deposits is approximately 7 m (Gaál and Gorbatschev, 1987). The national soil classification of Finland is based on texture and organic matter. The soil parent materials have been separated into three groups: till or moraine, sorted mineral soils (gravel, sand, fine sand, silt and clay) and organic soils. Most of the Finnish soil parent materials are classified as tills. Silt and clay exists mainly along the southern and western coastline. According to the revised FAO/UNESCO and WRB systems, Cambisols are most frequent and examples of Podzols,

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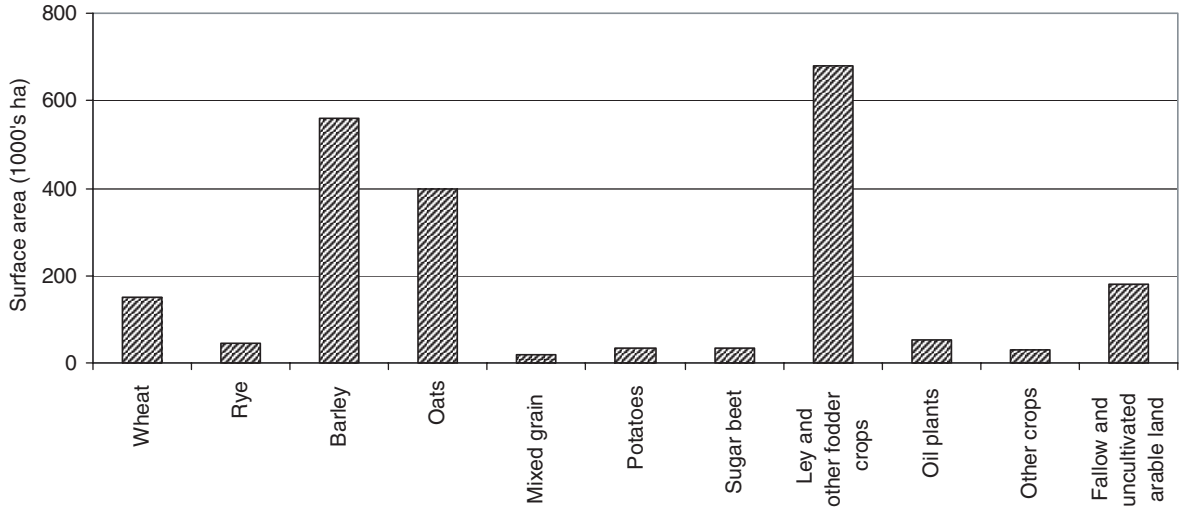

Figure 1.3.1 Use of arable land by various agricultural crops in Finland in 2000 (TIKE, 2000)

Histosols, Regosols, Gleysols, Arenosols, Umbrisols (WRB) and Phaeozems were identified (Yli-Halla *et al*., 2000).

1.3.3 AGRICULTURE

Between 1995 and 2000, the number of farms decreased at a rate of 3.5% per year, whereas agricultural land use remained fairly stable. In 2000, the area under cultivation and set-aside in Finland totalled 2.18 million ha, and the share of set-aside area was 0.2 million ha. Oats and barley account for about half (44%) of the area under arable crops, and the share of ley fodder crops is about one third (31%). Fallow and uncultivated arable land covers 182 000 ha (8%) of the arable land area (Figure 1.3.1) (TIKE, 2000). Compared with many regions in Europe, the amount of permanent grassland in Finland is small.

1.3.4 ON- AND OFF-SITE IMPACTS AND EROSION IN FINLAND

Erosion rates in Finland vary greatly owing to local natural conditions and management practices. In Finland, the highest erosion risk is in the south-western coastal area (Mansikkaniemi, 1982; Puustinen, 1994), although there are steeper slopes in central Finland (Figure 1.3.2). The risk there, however, is reduced by coarser soil texture and higher proportion of perennial grass and hay crops. Agricultural field plots and catchments dominated by agriculture (A1, A2, A3, A5) produce higher erosion rates than forested or mixed catchments (A4, A6, A7) (Table 1.3.1). As verified by numerous earlier studies, the smaller the test area, the higher are the mean erosion values and the larger is the range of variation (Table 1.3.1; A1 and A5).

Most erosion studies in Finland have been performed on the south-western coastal plains. This is due to three factors. First, the clayey soil type that is predominant in this area is more susceptible to erosion than the inland predominantly sandy and morainic soils. Second, the rivers flowing to coastal waters in south-western Finland have catchments with a relatively high field percentage (17–43%) compared with inland catchments.

Figure 1.3.2 Erosion risk areas in Finland and locations of erosion research plots and small catchments

Third, the coastal plain drains directly via rivers to the coastal waters, in contrast to the inland areas that drain via a complex system of lakes, rapids and rivers.

Typical erosion processes in Finland are sheet erosion, rill erosion and tillage erosion. During and after the period of snowmelt, rill erosion is the dominant process. Erosion generally occurs when surface runoff takes place. Erosion rates are not, however, linearly related to runoff rates. Water will, accumulate, for example, in depressions and is not necessarily seen immediately in erosion and surface runoff data. Similarly, the deposition of eroded material may occur locally when the transport capacity is less than the sediment load and thus diminish the cumulative erosion curve. The ratio of winter to total erosion varies from year to year, being mostly greater than 0.5. This ratio depends somewhat on the applied cultivation practice. For example, stubble cultivation enhances winter erosion. It also appears that the timing of the enhanced erosion and runoff in mid-spring seems to match with the time of the maximum snowmelting period. The effects of 10-m wide vegetative filter strips on sediment and nutrient losses from cropped soil plots have been studied at MTT Agrifood Research Finland. The filter strips decreased loads of total solids, phosphorus and nitrogen by an average of 23.6 and 47%, respectively. The grass buffer strips were effective in autumn but not in spring (Uusi-Kämppä and Kilpinen, 2000).

TABLE 1.3.1 Annual erosion amounts in Finland

Station name	Abbrev.	Date of start	Area (km^2)	Slope (%)	Soil type	Erosion range ($t\,ha^{-1}yr^{-1}$)
Agricultural field plots						
Aurajoki	A1	01/1988	0.0102	7–8	Clay	0.6–3.3 (winter wheat)
Kotkanoja	A2	01/1991	0.017	2	Clay	0.03–0.67 (winter wheat)

Catchment Name	Abbrev.	Date of start	Area (km^2)	Arable land (%)	Slope (%)	Soil type	Erosion range ($t\,ha^{-1}\,yr^{-1}$)
Catchments							
Hovi	A3	01/1981	0.12	100	2.8	Clay	0.1–2.35
Yli-Knuuttila	A4	01/1981	0.07	0	16	Moraines	0.021–0.256
Savijoki	A5	01/1981	15.4	39	4.8	Clay, moraines	0.082–0.646
Latosuonoja	A6	01/1981	5.34	19	8.2	Moraines, peat	0.015–0.104
Myllypuro	A7	01/1981	9.86	2	7.4	Moraines, peat	0.003–0.017

Data sources: Puustinen, 1994; Turtola, 1999; and long-term monitoring data of the Finnish Environment Institute (unpublished)

The primary concern with erosion on arable land in Finland is connected to off-site impacts of erosion (Figure 1.3.3). Soil erosion is a carrier of nutrients, particularly phosphorus, to surface waters, where it accelerates primary production resulting in eutrophication problems. Although dissolved phosphorus is mostly used by primary producers, particulate phosphorus losses also need to be reduced, because certain amounts can also be utilized by algae (Uusi-Kämppä *et al*., 2000; Uusitalo *et al*., 2001). Since the control of point-source pollution is well developed, e.g. phosphorus removal in waste water treatment plants currently exceeds 95% in Finland, the main focus of water conservation policy has been on diffuse pollution during recent years. Many of the control mechanisms are connected with erosion control.

Societal responses for erosion control, and more generally for pollution from agriculture, are monetary incentives. Farmers are reimbursed for leaving a vegetative filter strip between their fields and waterways (ditches, rivers and lakes), for increasing the share of vegetative cover during winter by avoiding and replacing autumn ploughing by mouldboard with more reduced tillage techniques and for establishing artificial ponds and wetlands to trap soil particles.

In addition to eutrophication, erosion also increases turbidity and silting of river beds, limiting their suitability for use, e.g. fishing and recreation. Most aspects of damage caused by erosion are difficult to measure and their financial value is hard to assess. Since a new national target was set to decrease the load of nutrients by about 50% by 2005, there is a need to decrease erosion as a nutrient carrier. The studies performed indicate that the impaired water quality might, at least locally, be a public nuisance and have economic consequences. In order to estimate the ecological consequences of off-site impacts, further work is still needed to quantify the effect and to link the nutrient load from agriculture to the eutrophication potential.

1.3.5 ONGOING SOIL EROSION STUDIES

Routine measurements of sediment load in small hydrological basins are carried out by SYKE (Finnish Environment Institute) and Regional Environment Centres. In order to study the effect of cultivation practices on soil erosion and nutrient transport, plot studies were established by MTT Agrifood Research Finland and the Finnish Environment Institute in the late 1980s and are ongoing (Puustinen, 1994; Turtola, 1999;

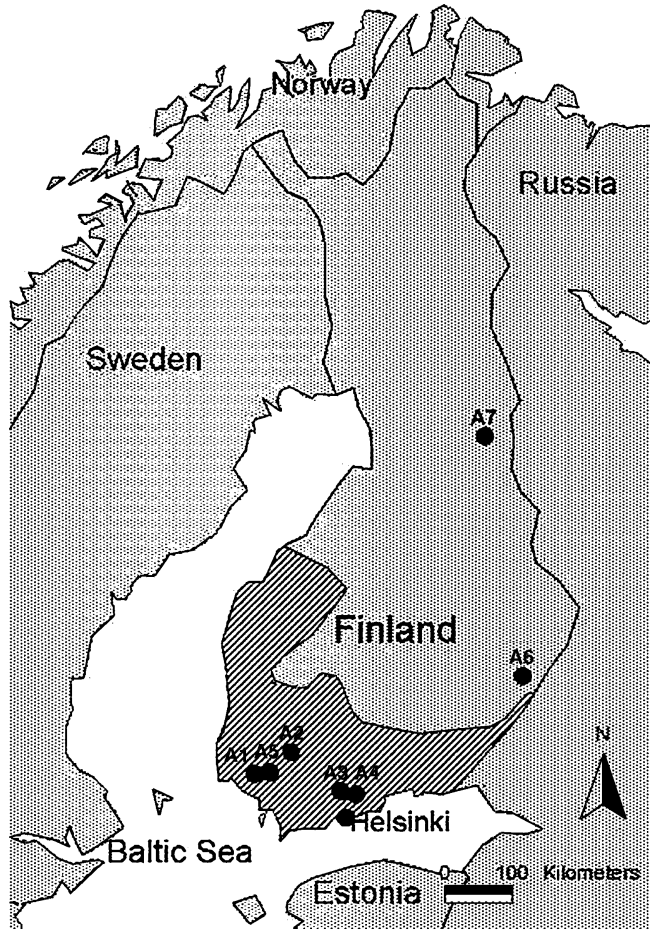

Figure 1.3.3 The area where erosion and nutrient load have deteriorated the quality of lake and river waters in Finland

Koskiaho *et al*., 2002). According to Puustinen *et al*. (2005), it is possible to decrease the amount of total suspended solids and phosphorus concentration by (i) reducing soil tillage, (ii) changing the tillage from autumn to spring or (iii) maintaining a permanent vegetation cover on the field surface. The effect gradually increases with the transition from intensive autumn tillage towards less intensive tillage practices and permanent vegetation cover. Erosion modelling (ICECREAM = Finnish version of CREAMS/GLEAMS models and SWAT) is carried out both in SYKE and MTT associated with nutrient transport modelling (Posch and Rekolainen, 1993; Rankinen *et al*., 2001; Tattari and Bärlund, 2001; Tattari *et al*., 2001). In addition, long-term snow water equivalent data are analysed in order to classify soil erosion events based on these data. This programme is part of the VIHMA (Management of Runoff Waters) project, which was initiated in 2002.

1.3.6 CONCLUSION

In Finland, the loss of eroded material and nutrients is highly dependent on the hydrological cycle. The actual effect of snowmelt and rain induced erosion on the annual sediment loss and the efficiency of different

management methods for reducing erosion and nutrient losses are currently studied based on long-term field experiments. Erosion rates vary between 0.03 and 3.3 t ha^{-1} yr^{-1} in agricultural areas, while the corresponding figures in forested catchments are considerably lower, namely 0.02–0.2 t ha^{-1} yr^{-1}. In Finland, the most noticeable effect of erosion is that eroded material carries nutrients, mainly phosphorus, causing eutrophication and harmful algal bloom in receiving waters.

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1.4

Denmark

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1.4.1 THE PHYSICAL ENVIRONMENT

The Danish landscape consists primarily of material of glacial and fluvioglacial origin from the Saale and Weichsel glaciations. The relief is low to moderate although steep slopes are found in the young moraine landscape with about 3% of the arable land being steeper than 10% and 1% steeper than 21% (Breuning Madsen *et al.*, 1987). Sandy soils characterize the western parts of the country (i.e. Central and Western Jutland), whereas the eastern parts are dominated by clayey till. The average yearly rainfall (1961–1990) is 712 mm, ranging from 900 mm in the western part to 550 mm in the eastern part. Rainfall values have not been corrected for aerodynamic and wetting losses which on average amount to 15% (Frich *et al.*, 1997). Erosivity calculated from daily rainfall observations is generally low, i.e. <10 and with maximum values around 35 (100 ft t acre^{-1} in^{-1} h^{-1}) (Leek and Olsen, 2000).

There has been a steady decline in the total agricultural area from 3094 km^2 in 1960 to 2647 km^2 in 2001 (Danish Bureau of Statistics, 2001). Part of this decline is attributed to the general increase in forested areas combined with an increase in the area covered by permanent grassland since 1994 (constituting 14% of the agricultural area in 2001). A governmentally driven afforestation programme has been an important driver in the increase in forested areas resulting from a strategy on the use of marginal land. There is hence a plan to double the forested area over a period of 80–100 years (Danish Forest and Nature Agency, 2000). Areas taken out of agricultural production consist of areas identified as being environmentally sensitive. Cereals are grown on 57% of the agricultural area, grass and green fodder in rotation on 16%, root crops on 4% and the remaining agricultural area is used for pulses, seeds for sowing, horticultural products and fruit trees (Danish Bureau of Statistics, 2001). Changes in land use have had a significant impact on soil erosion.

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TABLE 1.4.1 Types of water erosion processes in Denmark and typical soil erosion rates in t ha^{-1} yr^{-1}

Erosion type	Location/period	Plot size, slope and soil type	Land use	Erosion rate (t ha^{-1} yr^{-1})
Sheet erosion	Foulum and Ødum, 1989–92	22.1 × 3.0 m plots, 10%, sandy and clayey till	Bare soil	0.42
			Winter wheat across contours	0.95
			Winter wheat, contours	0.26
			Permanent ryegrass	0.03
			Spring barley followed by ryegrass during winter, ploughed in spring	0.13–0.42
			Spring barley, ploughed in autumn	0.45–2.69
			Winter wheat drilled up and down slope Ploughed in autumn	1.17–12.79
			Winter wheat drilled across the slope ploughed in autumn	0.49–11.08
			Fallow, ploughed in spring and harrowed from time to time to remove weeds	5.93–10.87
	Rabis and Syv, 1987–90	Catchment scale and plots (0.5 ha), 2–12%, sandy and clayey till	Catchments	0–0.14
			Plots	0–17.8
	Haraldsted, 2001–04	Catchment and plot (1 m^2)	Bare soil, winter wheat, spring barley (on-going research)	
Rill erosion	Rabis and Syv, 1987–90	Catchment scale and plots (0.5 ha), 2–12%, sandy and clayey till	Plots	4.25–19.6
			Slopes	0.35–18.6
	20 localities, 1993–99	Slope units, 2–20%, from sand to loam	Arable fields	0.47
	Gl. Lejre, 2000–01	Field size (5 ha), 15–20%, clayey till	Winter wheat	0.27
Tillage erosion	Sæby and Årslev, 1997–2000	Field size (1.5–8.0 ha), 2–19%, clayey till	Net erosion rate approx.	6.00
Bank erosion	Gjern river 1994–95	33 sites of 30 m stream reaches, 1.3–4.7 m wide, 22–61%, sand to loam	0.020 m^3 m^{-1} yr^{-1} stream reach	
	33 localities, 1998–99	Slope units 70–630 m long, 2.5–10%, sand to sandy loam	0.023 m^3 m^{-1} yr^{-1} stream reach	

Sources of data: (Djurhuus and Heckrath, 2000; Hansen, 1990; Hasholt, 1990, 1991, 1995; Heckrath, 2000; Kronvang *et al*. 1996, 2000b; Laubel *et al*. 1999, 2000; Schjønning *et al*. 1990; Sibbesen *et al*., 1994; Thers, 2001).

The most important processes of soil erosion in Denmark are wind erosion, sheet and rill erosion and bank erosion. Wind erosion has been known as a serious problem since the Viking Age and during the Iron Age wind drift caused several fields to be covered by sand. During the 17th century, several areas along the coast were covered by marine sand blown 10 km or more inland (Kuhlman, 1986; Knudsen and Vestergaard, 2001). There is no historical evidence of soil erosion caused by water. However, average annual soil loss rates from potentially erodible areas in Denmark are lower than in Norway where frost and snowmelt are some of the dominating factors combined with land levelling (Uhlen and Lundekvam, 1988; see also Chapter 1.1), but rates are similar to those found in the UK (Boardman, 1990) and Sweden (Alström and Bergman, 1990).

1.4.2 PROCESSES OF SOIL EROSION IN DENMARK

Five different types of soil erosion are identified in Denmark, namely wind erosion, sheet erosion, rill erosion, tillage erosion and bank erosion. Observed water erosion rates are listed in Table 1.4.1. Figure 1.4.1 shows the location of the study sites, except those not indicated by researchers. On-site effects caused by soil erosion are associated with wind erosion, whereas sheet, rill and bank erosion causes off-site problems in the form of eutrophication.

1.4.2.1 Wind Erosion

Wind erosion is a problem on sandy soils (0.1–0.5 mm) in western Denmark when sowing takes place. The finer particles (<1mm), including considerable amounts of organic matter, are removed preferentially from the surface. Figure 1.4.2 shows areas (altogether about 0.5 million ha) which in the 1960s experienced wind erosion. The same areas also experienced wind erosion in the 1970s and 1980s. The areas subject to wind

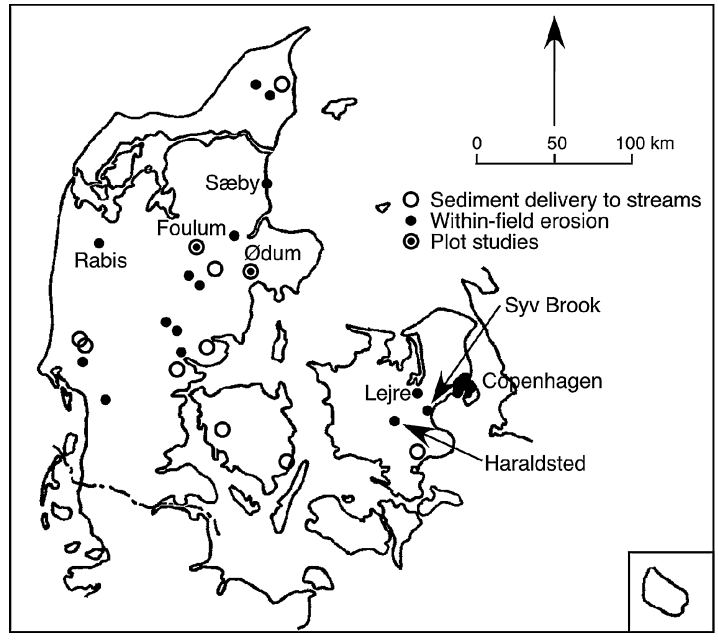

Figure 1.4.1 Location of major soil erosion research sites in Denmark

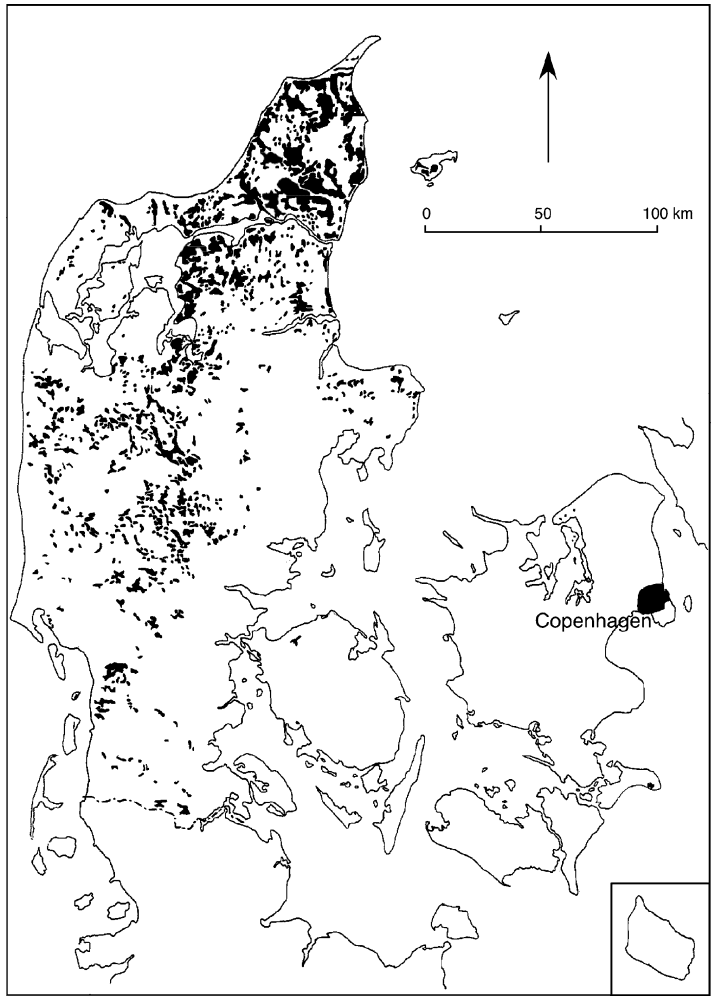

Figure 1.4.2 Major agricultural areas affected by wind erosion during the period 1960–70. (Modified after Kuhlman, 1986, with permission from the Royal Danish Geographical Society)

erosion are to a large extent identical with areas which, prior to cultivation, were described as heathland with sandy soils characterized by low soil fertility (Kuhlman, 1986). Although wind erosion to a large extent is under control, drought periods as experienced in spring 2003 still cause widespread wind erosion with economic consequences for farmers.

1.4.2.2 Sheet Erosion

Sheet erosion resulting in alluvial fan deposits is seen from time to time on most soil types in Denmark, most often in autumn and winter. The most severe cases are related to snowmelt and/or rainfall on partly frozen ground, but these cases also occur during high-intensity storms with saturated soil conditions, especially when ground cover is low. Fields with winter cereals are more prone to water erosion than ploughed fields as seedbed preparation promotes surface runoff and erosion by leveling the surface, compacting the soil and breaking

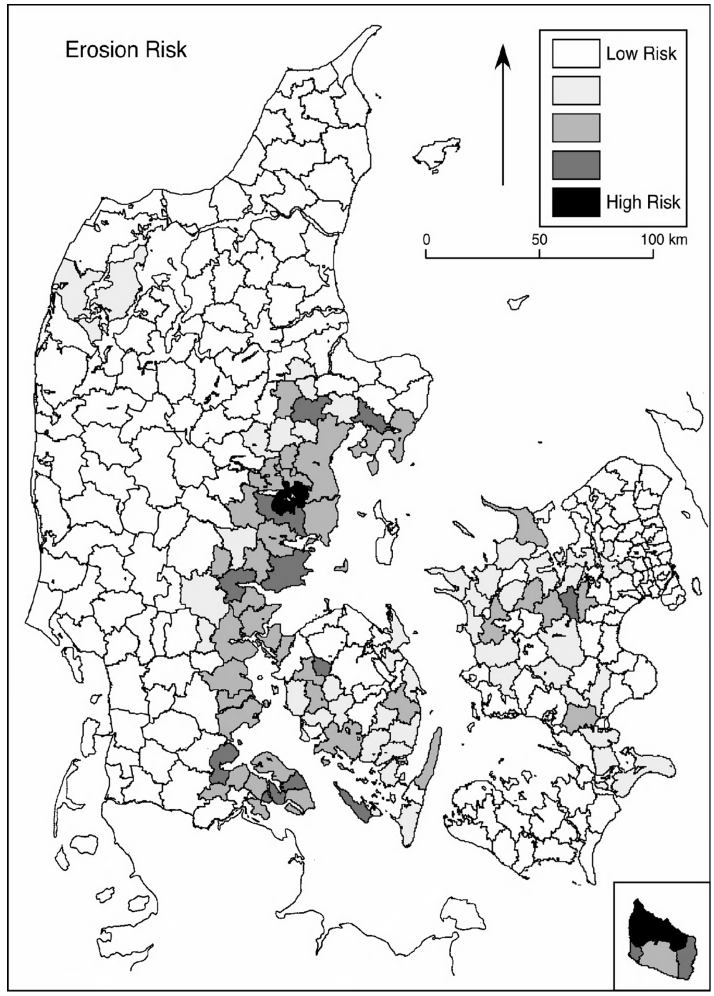

Figure 1.4.3 Areas of potential erosion risk in Denmark calculated with the USLE. (After Olsen and Kristensen, 1998, with kind permission of Springer Science and Business Media)

down soil structure. The sparse vegetation in winter-cereal fields during winter time also yields little shelter against the soil-slaking effect of raindrops, little resistance to surface runoff and has little soil-binding capacity (Sibbesen *et al*., 1994). The relative potential risk of soil erosion in Denmark calculated with the USLE is illustrated in Figure 1.4.3 and shows that the high-risk areas are located in eastern Denmark on soils developed on material from the last Weichsel glaciation consisting of till. These are also the areas primarily used for cereal production. However, the map has not been validated with regard to observed soil erosion sites since assessment of water erosion is not part of the standard agricultural monitoring programme.

1.4.2.3 Rill Erosion

Rills occur on all kinds of soils in Denmark. They are mainly formed on rather steep concave slopes, or below places where water can concentrate, such as roads or other areas with partly sealed surfaces. Clayey subsoil

seems to stimulate the formation of rills in the overlying soil. During summer and early autumn, rills mainly occur in connection with thunderstorms. The most prone areas during this time of year are areas with row crops, newly tilled areas or areas with sparse vegetation cover. Recently ploughed and subsequently harrowed fields are particularly prone to erosion, because of their low roughness and lack of protective vegetation. In autumn and winter, rill erosion typically occurs during four types of events: first by prolonged rainfall on moist soil, second in situations with rain on partly frozen soil, third during snowmelt events and fourth during high-intensity storms combined with low plant cover (Hasholt and Breuning-Madsen, 1989; Thers, 2001). The most severe cases of rill erosion are found on fields with winter wheat and barley, in particular in combination with clayey subsoils. Some of the rills are large enough to be classified as ephemeral gullies, whereas larger gullies typically found in loess areas are not found in Denmark.

1.4.2.4 Tillage Erosion

Tillage erosion has not been recognized in Denmark until recently (Djurhuus and Heckrath, 2000). The presence of this type of erosion has probably been increasing because of the use of heavier machinery and more efficient plough types. In steeper areas, this type of erosion is indicated by terrace-like steps, of up to 2 m between single fields.

Tillage erosion rates can be relatively high (Table 1.4.1), but this type of erosion is less important for long-distance transport of sediments and nutrients, as shown by Heckrath (2000) in Denmark. Harrowing of fallow plots favours tillage erosion by lowering the capacity of the surface soil for water assimilation and storage of ponded water (Sibbesen *et al*., 1994). A special type of tillage erosion is the removal of soil while harvesting sugar beet and potatoes. A denudation rate of 0.1 mm yr^{-1} from a single field has been observed based on the amount of sediment that is washed off the beet and potatoes at the factory (Hasholt, 1983).

1.4.2.5 Bank Erosion

Studies in Danish lowland catchments have shown that stream bank and bed erosion are very important processes, accounting for 40–80% of suspended sediment export with the largest contribution coming from the lower parts of the bank (Hasholt, 1988; Laubel *et al.*, 1999, 2000). There is considerable variation in bank erosion rates depending on site-specific factors such as soil type, bank vegetation, bank angle, bank height, stream power, stream form, buffer zone width and land use on the adjacent fields (Laubel *et al.*, 1999, 2000). Average bank erosion rates in the Gjern stream system in central Jutland was about 11 mm yr^{-1} per stream bank over a 1-year period, corresponding to 0.020 m^3 m^{-1} stream reach, and was generally found to be lower in forest streams than in streams on grassland used for grazing cattle (Laubel *et al.*, 1999). Similar studies along 15 small Danish lowland streams representative of the Danish landscape types revealed mean geometric bank erosion rates over the 11-month measuring period of 2.7 mm, equivalent to 0.023 m^3 m^{-1} stream reach (one streamside only). Erosion rates were significantly lower for sites adjacent to uncultivated areas than for those adjacent to agricultural fields and for sandy soils as compared with loamy sites (Laubel *et al.*, 2000).

Studies have shown that water erosion is an important factor in bank erosion, which leads to a slow undercutting of the bank, making the bank subject to failure (Laubel *et al.*, 2000; Obdrzalkova and Hovorkova, 2004). During a 2-year study of bank erosion, Laubel *et al.* (2003) found that erosion rates were related to site-specific characteristics including bank angle, bank vegetation cover, overhanging bank and estimated stream power. An empirical model for bank erosion based on these descriptive variables yielded a 55% explanation of the observed spatial variation in bank erosion rate. Process-based studies of bank failure and the effect of vegetation in particular are currently being carried out in southern Zealand as part of a research project by Roskilde University. Initial findings have shown large variations in vegetation cover on

banks. The convex part of the bank is covered primarily by grasses, whereas the concave parts are dominated by herbaceous plants. Although root densities (cm root cm^{-3}) are similar to those found on grassland and on fields with spring barley and although root densities more than 10 cm below the surface at the lower part of the bank are higher compared with the remaining part of the bank, bank failure processes are still visible. This indicates that pore pressure and soil saturation also are important factors with respect to bank failure.

Accurate estimates of bank erosion rates are difficult owing to large spatial variations in erosion/deposition rates. Laubel *et al.* (2000) found a coefficient of variation of 156% for erosion pins within a pin group, whereas studies by Obdrzalkova and Hovorkova (2004) showed coefficient of variation values ranging between 54 and 141%. Obdrzalkova and Hovorkova (2004) observed a highly dynamic erosion/deposition environment on the bank when using photo-electronic erosion pins (PEEP sensors), which demonstrates the importance of long-term continuous monitoring of bank erosion.

1.4.3 SOIL CONSERVATION AND POLICY MEASURES TO ADDRESS SOIL EROSION

1.4.3.1 Wind Erosion

In order to reduce the risk of wind erosion, soils are ploughed in winter and are compacted while wet. Seedbeds are prepared by harrowing to 3–5 cm depth at low speed and are left as rough as possible. In many cases, drilling and harrowing are done using combined equipment (Hansen, 1989).

Another measure used to reduce wind erosion is the establishment of windbreaks. The planting of windbreaks in Denmark can be described as a success story. This has been attributed to key elements such as farmer participation, good products and the involvement of the government. The government has provided subsidies since the 1880s, but it was only in 1976 that it passed a law specifically on windbreaks, which has since been revised several times (De danske plantningsforeninger, 2001; Knudsen and Vestergaard, 2001). From 2001 to 2002, the amount of subsidies given to planting societies (consisting of a group of farmers) was differentiated with larger subsidies to ecological farmers (45% as opposed to 40%) and to farmers who provided access to the public by establishing footpaths (50% subsidy) (De danske plantningsforeninger, 2001; Veihe *et al*., 2003). Following the change of government at the end of 2001, this differentiation is no longer made and a flat rate of 40% is given. Apart from the law specifically dealing with windbreaks, there are a number of other laws and notifications that influence the establishment and maintenance of windbreaks. These relate to the division of land in general, the improvement of flora and fauna in the biotopes (which include windbreaks), the protection of areas next to lakes, rivers and coasts, archaeological sites and any local decisions relating to road crossings (visibility requirements) (Knudsen and Vestergaard, 2001).

1.4.3.2 Water Erosion

Buffer zones have been found to be very efficient in terms of retaining sediment and phosphorus associated with rill erosion. Experiments have shown that all sediment and phosphorus are retained within the first 12 m of a 27-m buffer zone where the slope gradient is 14%, but slope gradient has a large effect on the trapping efficiency (Kronvang *et al*., 2000b).

Policy measures to address water erosion in Denmark are mainly done through the designation of ‘Specifically Vulnerable Agricultural Areas’ at the county level. Measures taken consist of set-aside, the use of rye catch crop and 2-m wide buffer strips around all water courses to prevent bank erosion caused by heavy machinery (Sibbesen and Iversen, 1997). Following the initiation of the Water Environmental

Protection Plans (WEPPs) in 1987, a green field strategy has been introduced by law to reduce nitrate leaching. This has had some unwanted side-effects in terms of increased soil erosion due to seedbed preparation during periods with excess rainfall causing increased surface erosion (Sibbesen *et al*., 1994; Hasholt *et al*., 1997).

1.4.4 CONCLUSIONS AND FUTURE PERSPECTIVES

The most important on-site impacts of soil erosion are damage to seeds and plants and the loss of organic material due to wind erosion. On the other hand, water erosion processes are associated with eutrophication problems, of which sheet, rill and bank erosion are the most important processes. Whereas wind erosion to a large extent is being controlled through soil management practices and the establishment of windbreaks, the off-site problems associated with water erosion have not been solved.

Recent assessments of eutrophication in coastal areas have shown that no significant improvement has taken place since the initial monitoring started in 1989. The water quality is not acceptable and a reduction in nutrients released through agriculture is necessary if the set targets for water quality are to be achieved (Danish Environmental Protection Agency, 2000; Ærtebjerg *et al*., 2002). The total loss of phosphorus from agricultural areas to watercourses has been estimated to be 0.4–0.5 kg ha^{-1}. Although net input has been reduced from 15 kg ha^{-1} in 1985 to 11 kg ha^{-1} in 1999 for the country as a whole, there has been a surplus input of phosphorus to agricultural areas within the same period (Danish Environmental Protection Agency, 2000). This is primarily associated with animal husbandry farms (Danish Environmental Protection Agency, 2000; Grant *et al*., 2000). Studies by Rubæk *et al*. (2001) from 1986 to 1997–98 have shown a yearly increase in phosphorus within the top 50 cm of 25 kg P ha^{-1} on average, although the increase is mainly on sandy soils where the number of animals per hectare is highest. There is consequently a high potential for future eutrophication problems in Danish watercourses and coastal areas if phosphorus is lost, a loss which has previously been underestimated (Grant *et al*., 2000; Kronvang *et al*., 2000a).

More research is needed on how frost influences soil erosion since this is one of the main parameters determining soil erosion processes in Denmark. There is an urgent need for modelling tools which can be used for identifying hot-spot areas within Denmark and to provide useful planning tools. The Danish Institute of Agricultural Sciences is currently working on the development of such a management tool. At the same time, proper validation of these modelling tools is urgently required. The long-term effect of soil erosion on soil fertility is also of major concern, especially in relation to tillage erosion and finally the development of better management tools for reducing bank erosion is needed.

Another challenge for the future will be to ensure that action plans and laws take into account the complexity of environmental issues, thereby avoiding unwanted side-effects. One example in Denmark is the use of winter cereals to reduce nitrate leaching. Although it has been documented that winter cereals are a main factor increasing soil erosion, the WEPP II stressed the demand for an increase in areas covered by winter crops (Danish Environmental Protection Agency, 2000).

An agreement was signed in April 2004 regarding an Action Plan for the Aquatic Environment III 2005–15, which focuses on the reduction of both nitrate and phosphorus loss. The Action Plan for the Aquatic Environment is closely related to the implementation of the EU Water Framework Directive and the Habitats Directive, which states that objectives and programmes of measures for individual water bodies and natural habitats to apply from 2009 must be laid down. Excess phosphorus should be reduced by 50% by 2015 through a tax of DKK 4 kg^{-1}on mineral phosphorus in feed and through general improvement of the phosphorus balance. During the same period, 50 km^2 of 10-m crop-free buffer zones along rivers and lakes will be established by voluntary set-aside land. This will be encouraged through subsidies. Furthermore, research into the mapping of risk areas in terms of phosphorus loss will be carried out.

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1.5

Iceland

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1.5.1 INTRODUCTION

Soil erosion and land degradation have reshaped the surface of Iceland. The consequences are severely damaged ecosystems, barren deserts and an unstable soil environment. Soil erosion remains extremely active in many areas of Iceland and the erosion landforms, such as the ''rofabards'', are among the main characteristics of Icelandic geomorphology.

It has been a national priority since the establishment of the Icelandic Soil Conservation Service in 1907 (''Landgraedsla rikisins'') to halt catastrophic erosion, especially encroaching sand, but later emphasis was directed towards reclaiming damaged areas.

This chapter gives a short overview of erosion in Iceland. It is based on a field survey of soil erosion in all of the country at a scale of 1:100 000, using a designated erosion assessment system, Landsat 5 satellite images as base maps and GIS systems for processing and storing the data. A comprehensive account of this work was published in English by Arnalds *et al*. (2001a) as a translation of earlier publication in Icelandic (Arnalds *et al*., 1997).

1.5.2 PHYSICAL GEOGRAPHY

1.5.2.1 Climate and Vegetation

Iceland is an island of about 103 000 km^2, located on the active North Atlantic Rift Zone. The interior of the island consists mostly of highland areas rising from 400 to >1000 m. Mountain ranges also extend to the shoreline in many areas, but lowland areas are situated along the coastline and river plains.

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Its northerly oceanic location, strongly influenced by the Gulf Stream, results in a cold temperate to subarctic climate with frequent freeze–thaw cycles during winter. The island is humid in most areas. Precipitation generally varies between 600 and 1500 mm yr^{-1} in lowland areas, but large tracts of north-east Iceland receive less than 600 mm. Much of the precipitation falls as snow in winter in north Iceland and in highlands, but winter thaw is common, especially in the southern part.

Classified satellite images (LMI, 1993) show that land with relatively continuous vegetation covers about 28 500 km^2, but an additional 23 900 km^2 has less continuous or nonproductive plant cover (a total of 52 400 km^2 of land with some vegetation to continuous vegetation). More than 37 000 km^2 is barren desert, some of which has formed after the settlement (874 AD). The vegetation composition of rangelands reflects sheep grazing, with species tolerant to grazing dominating most communities, such as small woody species and sedges. Lichens and mosses are characteristic of degraded land, but also of areas where succession takes place on new surfaces such as lava fields and disturbed land. Grasses are common, especially where grazing intensity is relatively low, or climatic/hydrological conditions are favourable. Herbaceous plants and some *Salix* species are indicators of moderate grazing. Birch woodlands used to cover a large proportion of the country but are now only about 1% (Aradottir and Arnalds, 2001). The barren surfaces are often sandy, consisting of volcanic glass and crystalline materials that are basaltic, colouring the surfaces dark or black. Almost all of Iceland was covered with glaciers during Quaternary glacial periods, but at the present time glaciers cover about 11 300 km^2 (LMI, 1993).

1.5.2.2 Soils

The formation of Icelandic soils is influenced by a steady flux of aeolian materials which originate from unstable desert surfaces. The rate of deposition commonly varies between 0.01 and 1 mm yr^{-1}, depending on distance from aeolian sources. Iceland has active volcanism, which affects the parent materials of the soils and permeability of the bedrock. Most regions are subjected to periodic ash-fall events during volcanic eruptions. The thickness of each layer is also variable, often 1–30 mm. Undisturbed soils of Iceland are primarily Andosols according to the WRB classification (FAO, 1998; Arnalds, 2004), which are soils that form in volcanic parent materials.

Soil drainage is also an important factor influencing Icelandic soils. Water permeability is rapid within the volcanic belt resulting in freely drained soils. Permeability is slower in the rock strata outside the belt of volcanic activity. This results in >22 000 km^2 of wetland soils which are chiefly Andosols, but Histosols are uncommon owing to the aeolian and tephra deposition which lowers the organic content of the wetland soils.

The soils of Icelandic deserts are termed Vitrisols in the Icelandic classification scheme (Arnalds, 2004), which include Andosols, Regosols and Arenosols according to the WRB. They consist of coarse-grained tephra materials, chiefly volcanic glass, but also varying amounts of clay minerals and some organic matter.

The properties of the Andosols are important in relation to the extensive erosion that takes place in Iceland. The soils are characterized by poorly crystalline clay minerals such as allophane and ferrihydrite, metal–humus complexes and considerable organic content. They are very friable and lack the cohesion that is usually provided by phyllosilicates in other soil types and many Icelandic soils exhibit thixotropic characteristics. These characteristics make the soils susceptible to erosion by water and slope failures. The formation of silt-sized aggregates is favoured, resulting in soils that are susceptible to erosion by wind (Arnalds *et al*., 1995). The Andosols can store large quantities of water, which aids water conservation and reduces erosion risk.

1.5.2.3 Agriculture and Land Use

About 290 000 people live in Iceland, mostly in towns, with only about 8% of the population living in rural areas (Statistics Iceland, 2002). Icelandic agriculture is primarily based on sheep farming and dairy

production, but poultry has recently gained ground. There are currently about 450 000 sheep (winter-fed ewes), 70 000 cattle (26 000 dairy cows) and 74 000 horses in Iceland (Farmers Association, 2003). Areas under crop production such as barley and vegetables are limited in extent. Hay for winter feeding of cows and sheep is grown on about 1220 km^2 of land (Farmers Association, 2003).

Sheep grazing is by far the most extensive land use, but in addition there is considerable horse grazing, mostly in lowland areas. Part of the sheep farming has relied on grazing of communal highland grazing areas. Many of the ecosystems that are being used for grazing by sheep can be considered 'marginal areas' because of vulnerable vegetation and soils, harsh climate and periodic volcanic ash-fall events. Overgrazing is still a problem in some areas in Iceland, and deserts and eroded areas that should not be used for grazing are still being used. It should be noted that extensive areas of Iceland can be considered suitable for sheep grazing and such land use is currently not causing extensive erosion problems, especially when compared with the problem areas.

Sheep production is currently facing various problems and the number of sheep is declining. This, ironical as it may seem, acts as an important factor in aiding in the recovery of Icelandic rangelands.

1.5.3 METHOD OF ASSESSMENT

1.5.3.1 General Characteristics of Erosion in Iceland

Erosion in Iceland occurs on rangelands. A distinction has to be made between erosion on desert areas, which lack vegetation cover for protection, and erosion associated with Andosols and vegetated ecosystems. A major characteristic of erosion of Andosols is that the entire soil mantle, often 50–150 cm thick, is removed by erosion processes, leaving the barren Vitrisol surface behind. Thus, Icelanders have most commonly assessed erosion by the loss of vegetative cover by hectare or percentage of vegetation cover lost. Erosion on deserts follows more conventional patterns, by both wind and water. Traditionally, erosion on deserts was only considered when it caused sand encroachment on vegetated areas.

1.5.3.2 Assessment Methods

Any method used for assessing soil erosion has to have clear objectives, which can be both scientific – a quest for better understanding of erosion processes, or the results are intended to have direct impact on how society reacts.

Most international methods for assessing soil erosion have been developed primarily for cultivated land, such as the Wind Erosion Equation, the Universal Soil Loss Equation and similar models. These models have proven to be very useful tools for both understanding erosion processes and predicting soil erosion problems. However, it can be argued that such methods often have limited applicability to grazing lands in mountainous areas. In Europe, noteworthy efforts have been made to map erosion risks (see Chapter 2.18), and the PESERA project seems to be promising for this purpose, for both cultivated land and grazing areas (Kirkby, 2003). It is not clear, however, how well these methods work as a baseline for regulatory frameworks for conserving soils, based on national law or locally driven participatory approaches. Other suggested or applied methods have focused on measuring various functions of the soil or ecosystem as a whole. Much used methods in the USA are based on evaluation of rangeland condition by sets of criteria that include both vegetation and soil erosion (e.g. NRCS, 1994). Such methods may well be better suited for assessment of open rangelands than conventional erosion models, but that is also dependent on the objectives of soil erosion assessments. Similar methods have been developed for evaluating the condition of grazing pastures in Iceland (Magnusson *et al.*, 1997), which have proved to be successful as a tool in participatory approaches to solve horse overgrazing problems.

1.5.4 NATIONAL EROSION SURVEY

The work on a National Soil Erosion Assessment was initiated in 1991. Field work was completed in 1996. The results were published in 1997 in a book entitled *Jardvegsrof a Islandi* or *Soil Erosion in Iceland* (Arnalds *et al*., 1997). The book includes both tables and maps for all of Iceland, regions, counties, municipalities and communal grazing areas. The results are stored in a GIS database which includes about 18 000 polygons with information about erosion types and severity. The information is considered 'public domain' and is distributed freely within Iceland. The project was awarded the Nordic Nature and Environmental Award in 1998. An English translation of the book was published in 2001. The English version does not include the detailed data for municipalities and commons. The book has also been translated into German and Danish, which will be posted on www.lbhi.is/desert, when these translations have been finalized.

1.5.4.1 Objectives of National Soil Erosion Assessment

Erosion in Iceland is a visible and publicized problem, and has been the subject of intense debate about the causes and the extent of erosion. Considerable resources are invested annually in halting erosion problems.

The objectives of the Icelandic National Soil Erosion Assessment were:

- to produce an overview of the soil erosion problem in Iceland, for land use decisions and planning and soil conservation strategies;
- to gain a better understanding of the main processes involved.

The assessment has had an important role in changing discussions from debates and a search for culprits, towards dialogue on solutions.

1.5.4.2 Methods

Methods for the assessment were developed in the light of the objectives stated above. The modes of soil erosion in Iceland vary considerably and it was evident that methods specific to Iceland had to be developed. The methods had also to take into consideration the great difference in erosion processes between the various parts of Iceland and between deserts and vegetated land. The conclusion was to base the mapping on erosion forms, with a view on site-specific differences. The method of separating erosion into erosion forms draws somewhat on methods developed in New Zealand (Eyles, 1985). Associated with each erosion form, a scale was developed to represent erosion severity. The erosion forms are listed in Table 1.5.1.

TABLE 1.5.1 The Icelandic erosion classification system (erosion forms)

Erosion forms associated with erosion of Andosols/Histosols	Desert erosion forms (Vitrisols)
Rofabards	Melar (lag gravel, till surfaces)
Advancing erosion fronts (sand encroachment)	Lavafield surfaces
Isolated spots	Sandur (bare sand, sand sources)
Isolated spots and solifluction	Sandy lava fields
features on slopes	Sandy melar (sandy lag gravel)
Water channels	Scree slopes
Landslides	Andosol remnants

TABLE 1.5.2 Erosion severity classes and land use policy of the Agricultural Research Institute and the Soil Conservation Service related to each class

Erosion class	Suggestions regarding grazing
0 No erosion	No suggestion
1 Little	No suggestion
2 Slight	Care needed
3 Considerable	Reduce or manage grazing
4 Severe	Protect – no grazing
5 Very severe	Protect – no grazing

The classification of erosion into erosion forms is in effect a geomorphologial approach to the problem, but the severity scale has a direct reference to land use decisions (Table 1.5.2). A policy statement by the Agricultural Research Institute and the Soil Conservation Service is built into the scale: no restrictions because of erosion are suggested for areas in low severity classes (0–2), but areas in erosion severity classes 4 and 5 are not considered suitable for grazing. Areas in erosion class 3 need further consideration and usually improvement. If such an area is a desert, it should not be grazed. The decision that grazing of Icelandic deserts is not an acceptable land use has been thoroughly explained in several documents (e.g. Arnalds *et al*., 2001a, 2003).

The assessment was carried out in the field by teams each consisting of two people. Erosion forms and severity were identified and marked on to Landsat 5 images and thereafter entered into an Arc/Info based database, using ILWIS-GIS for digitizing. The satellite images were used as geo-referenced base maps for Iceland. The mapping was done at the scale of 1:100 000.

1.5.4.3 Erosion Forms Associated with Erosion of Vegetated Land

Rofabards are perhaps the most distinctive erosion forms in Iceland (Figure 1.5.1). They were recently reviewed by Arnalds (2000). Rofabards are escarpments that range from about 20 to >3 m in height. They form in relative thick, noncohesive Andosols (mostly Gleyic and Brown Andosols), which overlie more cohesive materials such as glacial till and lava. The relatively loose Andosols beneath the root-mat is undermined, creating the escarpments. The rofabards retreat as a unit, with fully vegetated ecosystems on top, but leaving barren deserts in their place. Rofabards are common over an area of about 20 000 km^2, and the erosion database suggests that up to 15 000–20 000 km^2 of land that was previously fully vegetated and had fertile Andosols has now become desert as a result of the erosion processes associated with rofabards.

Many processes are active at rofabards, such as wind erosion, water erosion, gravitational processes (slumps), needle-ice formation and animal hoof impact. Lateral rain during high-intensity storms is an especially important factor in the high-rainfall areas of south and central Iceland, but wind erosion is more active in the drier areas of north Iceland.

Advancing fronts (encroaching sand) are called 'afoksgeirar' in Icelandic. They are active, tongue-shaped sandy surfaces extending into vegetated areas. These fronts start as sedimentary features (encroaching sand) that abrade and bury the vegetation with sand and destroy it. Sand fronts move into the vegetated land as the continuous flux of sand abrades the Andosol mantle and finally the new surface may be 1–2 m lower than the original surface. The advancing fronts are a major problem in Iceland that threaten fully vegetated systems, and they can advance over 300 m in a single year (Arnalds *et al.*, 2001b). Encroaching sand has desertified large areas in south and north-east Iceland, especially during the last part of the 19th century.

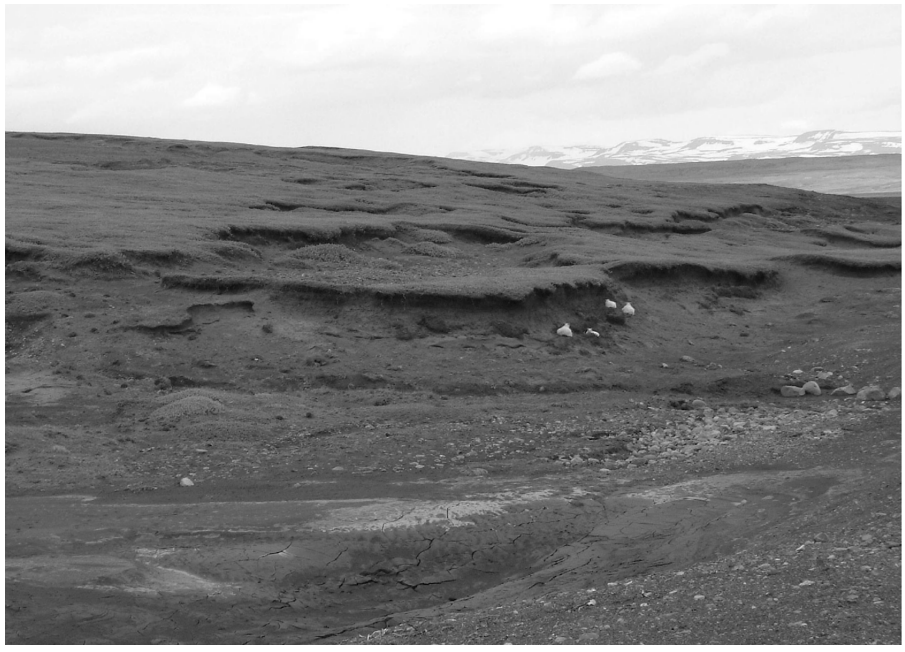

Figure 1.5.1 Icelandic rofabards. Sheep under the escarpment provide a scale

Isolated spots are small, bare patches in otherwise vegetated land. They are usually associated with hummocks, and are often a clear sign of overgrazing when they occur in lowland areas. Isolated spots are extremely common in Iceland, and their formation can lead to severe erosion (e.g. Gisladottir, 1998, 2001).

Isolated spots on slopes are a separate entity of the system. Erosion associated with such spots is more severe than on flat land, and commonly leads to slope failures. Solifluction is active on most slopes and, where those features are most pronounced (lobes and terraces), the danger of landslides is greater when isolated spots are dotting the landscape.

Landslides: during the mapping of erosion in Iceland, only landslides that occur on vegetated slopes were recorded. Such landslides are very common, hence the lack of stability of Icelandic Andosols.

1.5.4.4 Desert Erosion Forms

Deserts are divided into seven erosion forms based on geomorphology and stability of the surface.

Moldir are bare patches of Andosol remnants that often remain for some time after erosion has removed most of the soil. Their current aerial extent is low compared with other erosion forms.

Melur (glacial till or lag gravel surfaces) are usually surfaces that have lost their Andosol mantle because of erosion processes, but new melur surfaces also appear at the margins of receding glaciers. Some of the melur at highest elevations may never have accumulated much Andosol mantle. The surface of melur is subjected to erosion by wind and water and intense cryoturbation processes. The ground is often patterned and has a desert pavement surface.

Lavas are sparsely vegetated rock surfaces of the Holocene lavas that lack Andosol cover. Most often they are recent ($<$1000 years) or denuded surfaces by erosion processes. There is little erosion taking place on the lavas.

Figure 1.5.2 A sandur surface. This surface is almost totally devoid of vegetation and the surface is very unstable. Instrumentation being unstalled for relating sediment flux to climatic parameters. Other types of desert surfaces are more stable

Scree slopes are very common in mountainous areas. Many of these slopes may have been previously vegetated. Gravitational and water erosion processes are active on the slopes.

Sandur: the black basaltic desert sand-flats of Iceland are unique on a global scale (Figure 1.5.2). They are mostly formed by glacio-fluvial processes, during floods in glacial rivers or where glacial waters disappear into porous bedrock, leaving the sediments on the surface. Some of the sandflats have been formed by flow of a eolian materials from these sources. Sandur also includes sediments deposited during volcanic eruptions. These surfaces are extremely unstable and are subjected to severe and often spectacular wind erosion events. The sandy areas of Iceland were reviewed by Arnalds *et al*. (2001b).

The sandy materials are often moved by wind erosion and deposited over various desert surfaces. Two sandur surfaces represent such conditions: the *sandy melur* and the *sandy lavas*. The sand alters dramatically the conditions of the older surface as the sand is more unstable and has a low water-holding capacity. The sandy melur is the most widespread desert landform in Iceland.

Many of the sandy desert areas can be considered as natural desert areas and should be protected as such because of their uniqueness. Some of these areas should therefore be kept as set-aside land without reclamation attempts. Other land use that interferes with their natural development should also be restricted, such as grazing.

1.5.5 EROSION SEVERITY AND EXTENT

An overview of soil erosion in Iceland is presented in Table 1.5.3 and Figures 1.5.3 and 1.5.4. Severe and very severe erosion, which may be considered erosion hotspots in a European context, occurs on about 17% of

TABLE 1.5.3 Summary of soil erosion in Iceland

Erosion class	Area (km^2)	Proportion of country (%)
0 No erosion	4148	4.0
1 Little erosion	7466	7.3
2 Slight erosion	26698	26.0
3 Considerable erosion	23106	22.5
4 Severe erosion	11322	11.0
5 Extremely severe erosion	6375	6.2
Mountains	9794	9.5
Glaciers	11361	11.1
Rivers and lakes	1436	1.4
Unmapped	1010	1.0
Total	102721	100

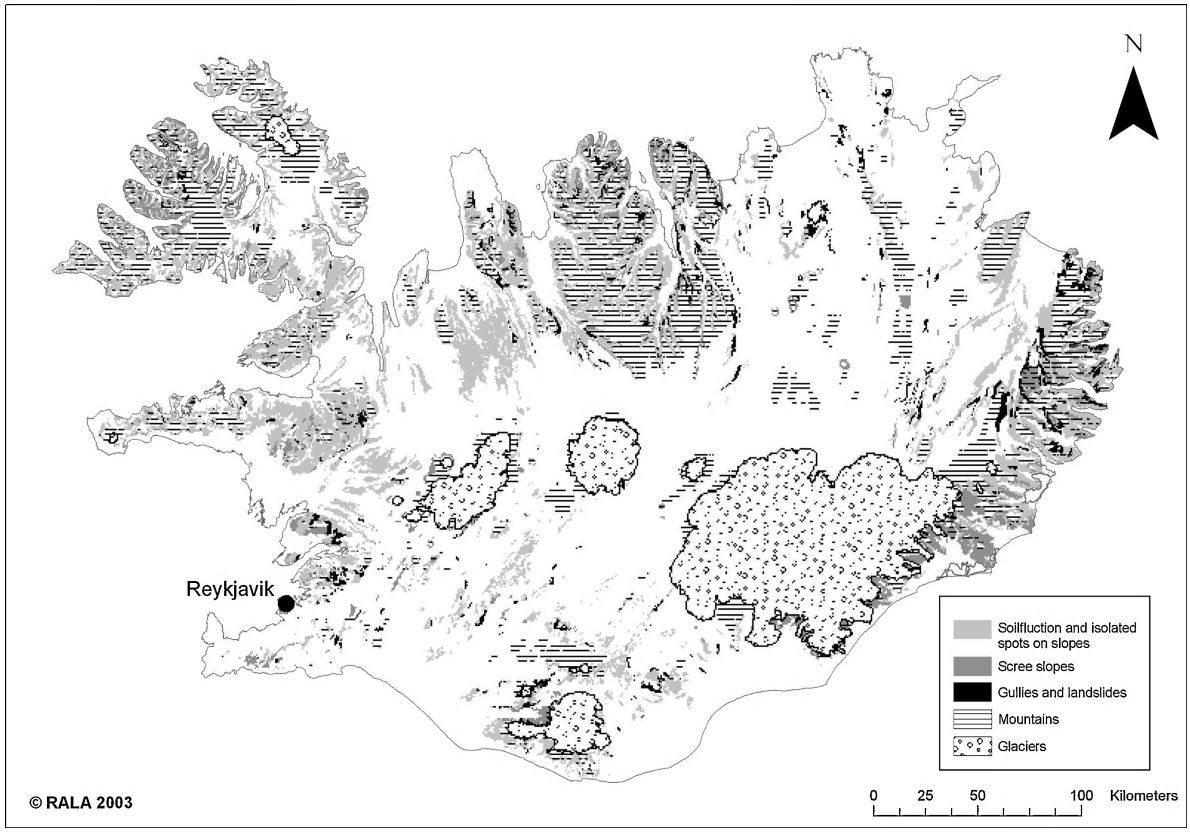

Figure 1.5.3 Soil erosion on hillslopes. Erosion forms include solifluction and isolated spots on slopes (severity classes 2–5), gullies and lanslides (2–5) and scree slopes (3–5)

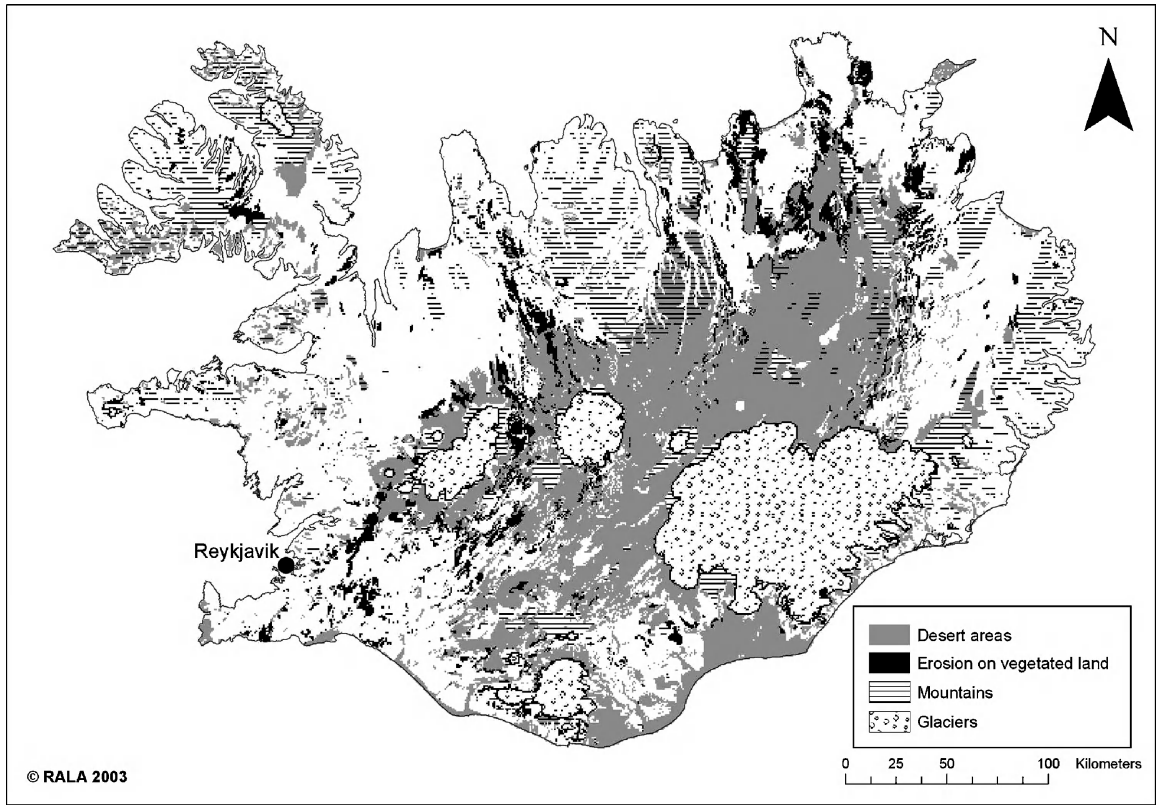

Figure 1.5.4 Soil erosion on relatively level land. The desert areas include melur, sandur, sandy lavas and sandy melur (erosion severity classes 3–5). Some of the desert areas include hillslopes. The erosion forms associated with vegetated land include rofabards and isolated spots (severity classes 3–5)

Iceland. Considerable erosion (severity class 3) occurs on 22% of Iceland and therefore erosion can be considered a substantial problem on 40% of Iceland (classes 3–5), or about half of the country when glaciers, water bodies and high mountains are excluded. These results are of great concern, and are considered in the National Soil Conservation Strategy discussed in the policy section of this paper.

The results are given in greater detail in Table 1.5.4, by erosion forms and erosion severity. The table indicates that rofabards occur on an area of about 8800 km^2 and is classified as considerable on nearly 2000 km^2, severe on 1234 km^2 and very severe on 361 km^2. This aerial extent is lower than most would expect, as these are very prominent features on Icelandic landscapes. There is evidence that this area may have been much larger during the past 1000 years (Arnalds, 2000). Advancing sand fronts are only delineated around the area where the sand is encroaching on vegetation and the aerial extent is therefore limited (86 km^2), but the erosion threat is great in these areas. Isolated spots are very common (>28 000 km^2), but most of this erosion is not pronounced (mostly severity classes 1 and 2). However, the 2729 km^2 area of isolated spots in severity class 3 is noteworthy, and very commonly represents fully vegetated areas that are being overgrazed. The same applies to isolated spots on hillslopes in active solifluction areas and the results indicate that extra steps need to be taken to protect soils on hillslopes, and especially to halt the current increase in horse grazing on these slopes. Gullies and landslides occur on much smaller areas.

TABLE 1.5.4 Division of the country according to erosion forms and erosion classes (in km^2)[a]

	Erosion class					
Erosion form	1	2	3	4	5	Total
Rofabards	1735	3511	1997	1234	361	8837
Encroaching sand	2	4	13	40	26	86
Isolated spots	6929	18456	2729	103	0	28217
Solifluction/spots on slopes	924	10702	5962	109	1	17697
Landslides	398	190	89	6	0	683
Gullies	740	2572	1236	107	42	4652
Melur	9939	8546	6580	0	0	25065
Lava	1832	228	25	0	0	2085
Sand	195	337	318	1087	2828	4765
Sandy gravel	8	741	5407	6217	1286	13659
Sandy lava	10	101	1366	1757	1620	4855
Soil remnants	17	518	350	65	36	987
Scree	64	913	2,378	1,255	392	5002
Total	22794	46775	28449	11979	6595	116592

[a]Note that many polygons are counted more than once (multiple erosion forms within the same polygon), which is why the total land area is large. Mountains, glaciers, rivers, lakes and unmapped areas are excluded from the calculations.

Melur is the most common desert land form, but most often associated with vegetated patches where it receives a severity class lower than 3. It is evident from Table 1.5.4 that severe and very severe erosion occurs primarily on the sandy deserts.

1.5.6 HISTORICAL NOTES

There is clear evidence for a dramatic environmental change at the time of settlement by Nordic Vikings during the 9th century. After settlement, rapid population growth led to intensive use of fragile ecosystems. Vegetation changes were pronounced and erosion escalated. This is shown by a 4–10-fold increase in aeolian deposition rates at that time (Thorarinsson, 1961). This has resulted in a thicker Andosol mantle, which is more vulnerable to erosion than the previous surface. Evidence for past changes includes historical records, Sagas, annals, old farm surveys, old place names, relict areas and current vegetation remnants, pollen analyses and soils buried under sand (e.g. Thorarinsson, 1961; Arnalds, 1987, 1988; Hallsdottir, 1995; Kristinsson, 1995; Gisladottir, 1998; Dugmore *et al*., 2000). There is no documented evidence for such massive country-wide erosion in Iceland before the settlement.

The causes of the dramatic erosion have traditionally been attributed to human pressure, but many other factors also contribute to the degradation of Icelandic ecosystems. Climate was already becoming cooler at the time of settlement, a trend that started about 2500 years ago. This made some of the marginal ecosystems very vulnerable to disturbance. It has been suggested that some of the observed changes at high elevations may be attributed to climate change alone (Olafsdottir *et al*., 2001). This cooling trend has undoubtedly increased the size of glaciers and the size of active aeolian deserts at their margin with increased number of melt-water floods. Frequent episodes of volcanic ash deposition and cold spells, particularly between 1400 and 1800, have also had an escalating effect on erosion that had already began. It is most likely, however, that in many cases

the land use triggered a snowball effect, escalated by volcanic ash deposition, cold spells and ever growing human pressure.

1.5.7 SOIL CONSERVATION PRACTICES

The Icelandic Soil Conservation Service (SCS) ('Landgraedsla rikisins') was formed in 1907, representing one of the oldest operating government institutes of its kind. The first objective of the SCS was to halt encroaching sand threatening rich vegetated systems. The history of soil conservation of Iceland has been reviewed by Runolfsson (1987), Magnusson (1997) and Aradottir (2003). The main emphasis of the Icelandic SCS was for a long time directed towards reclamation of severely degraded lands by application of fertilizers and seeding of grasses. It has also had a major role in monitoring and ensuring sustainable grazing practices, a role that has been increasing in importance. Since about 1985, the SCS has put steadily more emphasis on land-care projects and participatory approaches to reclamation work and for ensuring sustainable land use (Arnalds, 1999). The project 'farmers reclaim the land' has been particularly effective in increasing land literacy. Such approaches do not, however, solve land use problems of deserts and erosion areas of the highland commons, where new operational law is needed (Arnalds and Barkarson, 2003).

The SCS has now implemented a 10-year Soil Conservation Strategy that is approved by the Parliament ('Althingi'). The strategy outlines objectives of the society in relation to soil projection and land reclamation. It emphasizes research and the professional skills needed. The erosion assessment is one of the backbones of the strategy. The context of sustainable development and the UN conventions on Climate Change, Desertification and Biodiversity is emphasized in the Soil Conservation Strategy. This context provides a new evolving paradigm for land reclamation, shifting from agronomic principles and practices towards ecological-oriented methodology (Aradottir, 2003).

An additional recent development is a noteworthy agreement between the government and sheep farmers, where part of the production subsidies are tied to 'quality management' that includes sustainable land use (Arnalds and Barkarson, 2003). Those farmers who meet a given land use criterion (in addition to good farming practices) will receive up to 22.5% higher payments than other sheep farmers.

Sheep grazing in the highland desert areas has very little economic significance and is important to only a small number of sheep farmers today. However, most of the poor-condition highland commons are still being grazed. The current law for soil protection and land reclamation was introduced in 1965. It is interesting that old laws, from the 12th and 13th centuries, had clear rules about sustainable grazing methods and the responsibility of animal owners to control their livestock. Today, each landowner has to fence off their land, in order to exclude sheep from their property, which involves high fencing costs. This is currently causing conflicts that are likely to escalate during the next few years.

1.5.8 CONCLUSIONS

Erosion is perhaps more active in Iceland than in any other European country. Natural conditions, the combined effect of such factors as soils, volcanic activity, land use and climate, differ from conditions in other parts of Europe, resulting in different erosion processes and landforms. The Icelandic National Soil Erosion Assessment is an example of country-specific methodology designed for local conditions and objectives. The assessment places Iceland in a different situation than most other European countries, with a detailed coverage of the erosion problems in the country. This view is based on field survey, but not on modelling of erosion/erosion risk or by assessment of erosion in parts of the country.

A complete soil erosion assessment has led to important changes in how society deals with the problems. Debates about the nature and extent of the problem have changed and the present discussion focuses more on solutions. Important steps towards more sustainable use of range resources in Iceland have recently been taken, partly as a result of the erosion assessment. Its methodology is now used on a regular basis for land assessment on farmland.

Lack of erosion assessment should not, however, prevent the development of laws for prohibiting land use that causes soil erosion or the facilitation of programmes such as land-care or participatory projects that increase land literacy and stewardship.

Soil erosion problems in Iceland are being addressed on several levels of society with the SCS as the national agency responsible for actions taken. Approaches include land-care programmes, a National Soil Conservation Strategy, subsidy-driven financial incentives and direct intervention to stop erosion. The Icelandic Soil Conservation Law is outdated, and a new one is needed that excludes land use on desert and erosion areas of the highland commons.

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1.6

Lithuania

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1.6.1 PHYSICAL GEOGRAPHY AND SOILS

Lithuania has a temperate climate, transitional between maritime and continental. Weather conditions are variable, with frequent winter frosts and cool, humid summers. The mean annual temperature is 6 °C; the January mean is −4.8 °C and July 17.2 °C. The climate is humid, with a mean annual precipitation of 675 mm. However, this is spatially variable, being highest (920 mm) in the south-west Zemaiciai Uplands and lowest (520 mm) in the northern Central Lithuanian Lowland. The Lithuanian climate is conducive to water erosion and heavy showers are particularly erosive. Heavy showers with >30 mm of rain occur in the Central Lithuanian Lowland about once every 2 years, in the south-west Zemaiciai Uplands about three times every 2 years and elsewhere about once per year. The mean wind velocity on the Baltic coast is 5.5–6.0 m s^{-1} and decreases to 2.9–3.5 m s^{-1} inland. In winter, owing to active cyclonic activity, wind velocities are 1–2 m s^{-1} greater than in summer (Arlauskiene *et al*., 2001).

Lithuania occupies the western fringe of the East European Plain and is predominantly a lowland country. These lowlands are separated by hilly uplands, forming two meridian-oriented stretches. The western edge of the Baltic Uplands is in the east and south of the Republic, where erosion processes affect large areas. The 'island-like' Zemaiciai Upland is in the west, where erosion processes affect 5.1–20 and 20–30% of the undulating terrain (Figure 1.6.1).

Moraines are the prevalent soil parent material, deposited in glacial margin and basal conditions. Ground moraine covers 30% of the national territory and glacial margin formations 27%. Glacio-lacustrine formations cover 23% and fluvioglacial formations 7%. Peaty, marine (littoral), aeolian and karst formations occupy only 0.2–1% (Arlauskiene *et al*., 2001).

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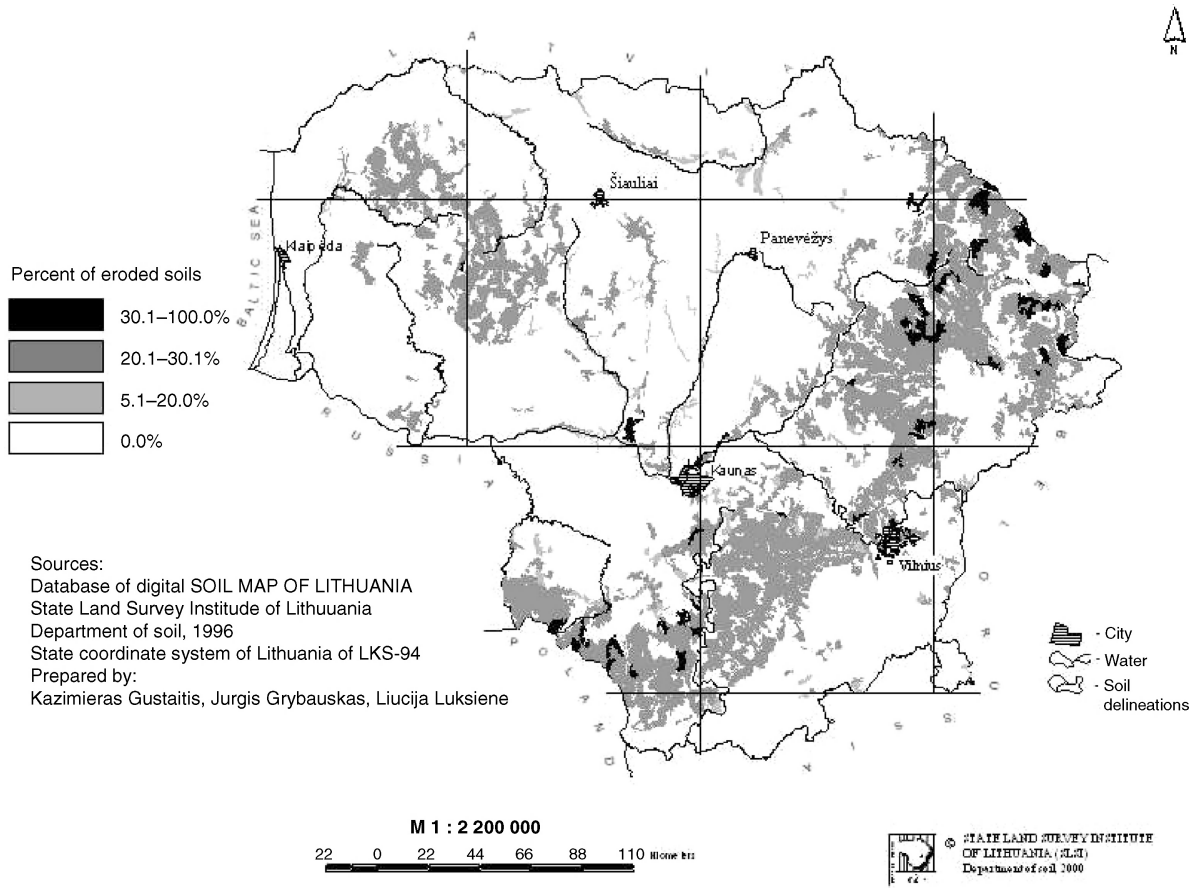

Figure 1.6.1 Map of eroded soils of Lithuania. (Reprinted from *NATO Science Series*, Vol. 44, Jankauskas B and Jankauskiene G, Ecological land use in the undulating landscape of Lithuania and Baltic sea environment, p. 206, copyright 2004, with permission of IOS Press)

About 52% of Lithuania's relief is undulating hills, where the soil is erodible (Kudaba, 1983) and ∼17% of Lithuania's agricultural land is eroded, increasing to 43–58% in the hilly regions (Jankauskas, 1996; Jankauskas and Fullen, 2002). The hilly rolling relief of Lithuania, dissected by gullies and river valleys, was formed in the early Holocene, after glacial melting ∼12 000 years BP (Baltrunas and Pukelyte, 1998).

1.6.2 HISTORICAL EROSION

The erodible glacial moraine, combined with the abundance and intensity of precipitation, created favourable conditions for water erosion in the early postglacial period. Agricultural activities commenced in Lithuania only at the end of the Neolithic, i.e. ∼5000 (Dunduliene, 1963) or ∼4000 (Gudelis, 1958) years BP. However, intensive husbandry and associated risks of soil erosion began in the 12th century, ∼800 years BP.

One representative transect from the group of 18 investigated longitudinal transects on the Zemaiciai Uplands is shown in Figure 1.6.2. Soil profile S0 was an uneroded profile in a wood. The calcareous soil

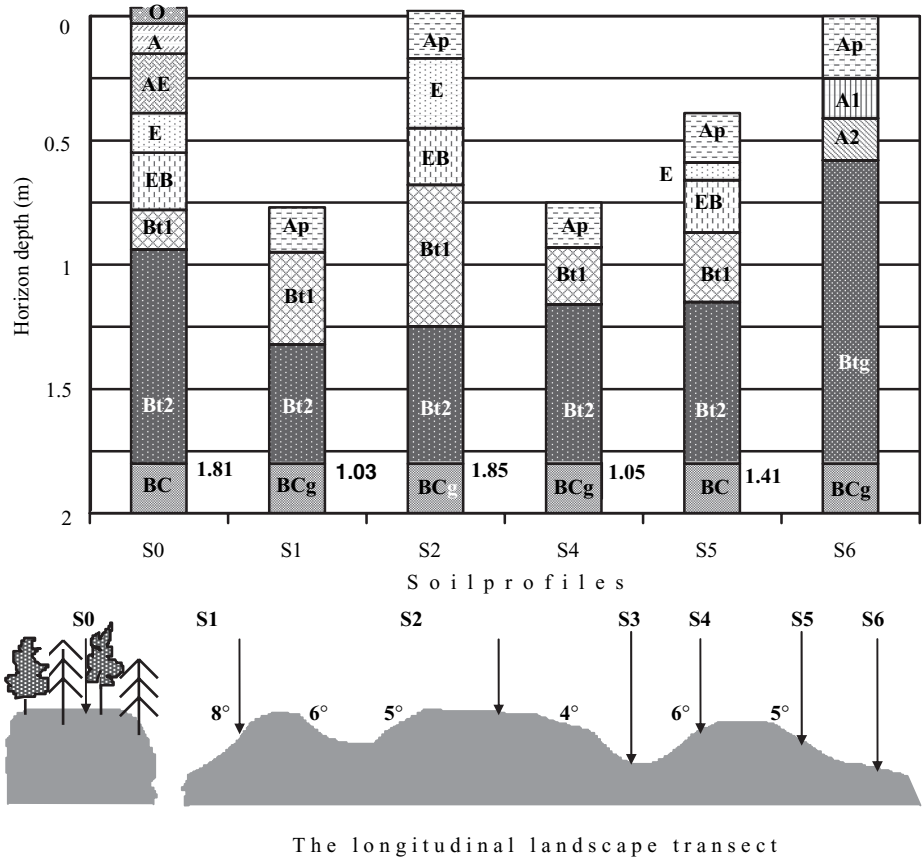

Figure 1.6.2 Severity of soil erosion on transect S. S0–S6, soil profiles: S0, noneroded soil in a woodland; S1 and S4, very severely eroded soil; S2, slightly eroded soil; S5, severely eroded soil; S6, colluvial soil on a foot-slope. Arrows indicate the locations of soil profiles. The white line indicates the depth of calcareous horizon and the adjacent numbers indicate depth (m)

horizon there was at 1.81 m depth and 1.85 m on soil profile S2 on the sloping plateau (Figure 1.6.2). This depth to the calcareous soil horizon was used as the basis for the calculation of eroded soil on the transect. The thickness of soil above the calcareous horizon was 1.03 m (soil profile S1), 1.05 m (soil profile S4) and 1.41 m (soil profile S5). Therefore, the estimated approximate thickness of lost (truncated) soil was 0.8 m on the 8° (13.9%) slope (soil profile S1), 0.8 m on the 6° (10%) slope (soil profile S4) and 0.4 m on the 5° (8.3%) slope (soil profile S5).

1.6.3 CURRENT EROSION

Soil erosion intensity in Lithuania depends mainly on tillage (mechanical) erosion, which has been identified as the main cause of accelerated soil erosion on arable slopes (Kiburys, 1989; Jankauskas, 1996). Agricultural implements (such as ploughs, cultivators and harrows) were used for tillage, which encouraged soil

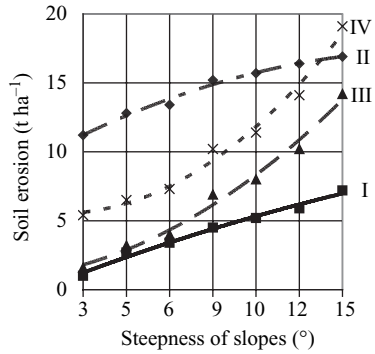

Figure 1.6.3 Dependence of tillage soil erosion on slope steepness after single mouldboard ploughing in different directions (Jankauskas and Kiburys, 2000). I, up and down slope; II, along the contour; III, slantwise across the slope in the right direction; IV, slantwise across the slope in the left direction

translocation on the hilly relief in the mid-20th century. Soil management systems became particularly intensive during the Soviet period. Therefore, investigations of tillage erosion processes were initiated at the Department of Agriculture of Vilnius Pedagogical University in 1960 (Kiburys, 1989). The rate of soil translocation under tillage erosion depends on slope steepness, tillage equipment and the direction of tillage operations. Farmers often create favourable conditions for water and wind erosion using tillage equipment on hilly relief. For example, the mass of soil moved downslope was 17.6 t ha^{-1} after a single mouldboard ploughing along the contour on a 100 m length and 10° (17.7%) slope and the mass of soil moved upslope was 1.9 t ha^{-1}. Therefore, the net rate of tillage erosion (difference between downslope and upslope movement, 17.6−1.9) was 15.7 t ha^{-1}. Tillage erosion was 11.4 t ha^{-1} when ploughing slantwise across the slope in the left direction and 8.0 t ha^{-1} when ploughing slantwise in the right direction. Tillage erosion was only 5.2 t ha^{-1} when ploughing up and down slope (Kiburys, 1989). Tillage erosion rates due to a single sequence of mouldboard ploughing on slopes from 3 to 15° (5 to 26.3%) were 1.0–7.2 t ha^{-1} when ploughing up and down slope, and 11.2–16.8 t ha^{-1} when ploughing across the slope (Figure 1.6.3).

According to the data presented in Figure 1.6.3, the relationship between slope steepness and tillage erosion can be expressed by the following equations:

$$y_{\mathrm{I}} = -0.09x^2 + 1.67x + 9.63, r^2 = 0.987,\ p < 0.05$$

$$y_{\mathrm{II}} = -0.03x^2 + 1.22x + 0.04, r^2 = 0.987,\ p < 0.05$$

$$y_{\mathrm{III}} = 0.18x^2 + 0.53x + 1.1, r^2 = 0.987,\ p < 0.01$$

$$y_{\mathrm{IV}} = 0.3x^2 - 0.28x + 5.6, r^2 = 0.986,\ p < 0.01$$

where y is soil losses (t ha^{-1}), x is slope inclination (°), $n = 10$.

Tillage erosion only moved soil over a short distance (75–85 cm), whereas water and wind erosion transported soil much further (Kiburys and Jankauskas, 1997). Therefore, formation of natural agro-terraces near natural or artificial boundaries is characteristic of arable hillslopes as a result of tillage erosion (Jankauskas and Kiburys, 2000).

Investigations of water erosion have been concentrated at the Kaltinenai and Dukstas Research Stations of the Lithuanian Institute of Agriculture. Both Stations were established in 1960. The oldest operational soil

erosion monitoring sites have been operated by the efforts of Dr A. Pajarskaite in 1960 at the Dukstas Research Station (Pajarskaite, 1965). There were monitoring sites with bare fallow, grain crops, grasses and wasteland (untilled/uncultivated land) from 1961 to 2002. The research data of the Dukstas Research Station represent soil and meteorological conditions in the Baltic Uplands. Runoff and losses of clay loam soil due to water erosion on the hillslopes of Eastern Lithuania ranged markedly, from 6.6 mm yr^{-1} of runoff water from wasteland to 151 mm yr^{-1} under bare fallow, or from 1.3 t ha^{-1} yr^{-1} of soil under cereal grain crops to 56.6 t ha^{-1} yr^{-1} under bare fallow on 5–7° (8.3–11.9%) slopes (Svedas, 1974; Bieliauskas, 1985).

Investigations of soil erosion on the Zemaiciai Uplands of western Lithuania at the Kaltinenai Research Station were initiated by Dr E. Cicelyte, and had been developed and comprehensively described by Dr O. Visockis. The physical and chemical properties of eroded soil were investigated and initial recommendations were made for soil conservation on arable slopes (Visockis, 1971). Evidence was presented that perennial grasses provided excellent protection against soil erosion, even on 10–15° (17.7–26.3%) slopes. Permanent legume–grass mixtures with a high percentage (90%) of common alfalfa (*Medicago sativa* L.) were more suitable for pastures on eroded slopes, if soils were suitable for growing alfalfa.

Requirements for other kinds of products (such as grain, tuber crops and root vegetables) encouraged investigations of crop rotations suitable for undulating hilly relief. Erosion-preventive 6-year crop rotations have been investigated on experimental plots at the Kaltinenai Research Station since 1983. Heavy losses of Eutric Albeluvisols (Aquic Glossoboralfs) occur owing to water erosion on the Zemaiciai Uplands under the field crop rotation (Jankauskas, 1996; Jankauskas and Jankauskiene, 2000). Study sites A, B and C were on slopes of 2–5, 5–10 and 10–14°, respectively (Figure 1.6.4). Field trial plot size was 338.4 m^2 (3.6 × 90 m) on sites A and C (slopes 2–5 and 10–14°) and 158.4 m^2 (3.6 × 40 m) on site B (slope 5–10°). On the long-term monitoring sites, the mean water erosion rate under the field crop rotation, containing 1 year of potatoes, 3 years of cereal grains and only two fields of grasses, was 23.4 t ha^{-1} yr^{-1} on the 5–10° (8.3–17.7%) slope. The rates increased with increasing slope inclination and were lower on the 2–5° (3.5–8.3%) slope. The erosion-protection capabilities of different crop rotations and land use systems varied widely. According to the mean data of 36 experiments (18 years of investigation on two blocks), the mean annual erosion rates under erosion-preventive grass–grain crop rotations decreased by 74.7–79.5% compared with the field crop rotation, containing 4 years of perennial grasses and 2 years of cereal grain crops. Under the grain–grass crop rotation, containing 4 years of cereal grains and 2 years of grasses, the rate decreased by 22.7–24.2% (Figure 1.6.4). However, even grass–grain crop rotations could not completely prevent water erosion, with mean rates of

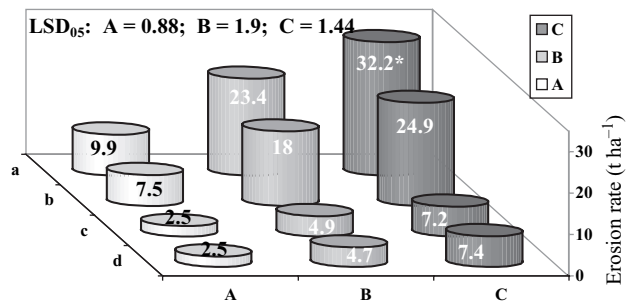

Figure 1.6.4 Annual water erosion rates under different crop rotations. The heights of columns represent the mean data for 1983–2000 on slopes: A, 2–5° (3.5–8.3%); B, 5–10° (8.3–17.7%); C, 10–14° (17.7–24.5%). (a) Field crop rotation; (b) grain–grass crop rotation; (c) grass–grain I crop rotation; (d) grass–grain II crop rotation. *The sod-forming perennial grasses were grown instead of the field crop rotation on the slope of 10–14°. Therefore, the water erosion rate for field crop rotation on the slope of 10–14° was calculated by the method of data group comparison

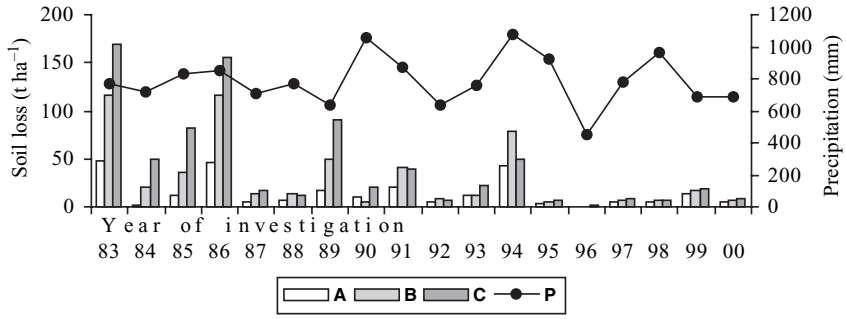

Figure 1.6.5 Soil losses from slopes of different gradient (columns) under spring barley; annual precipitation (line). Columns: slopes of A, 2–5° (3.5–8.3%); B, 5–10° (8.3–17.7%); and C, 10–14° (17.7–24.5%). P: total precipitation (mm)

7.2–7.4 t ha^{-1} yr^{-1} on the 10–14° (17.7–24.5%) slopes, which exceed tolerable levels (Fullen and Reed, 1986; Richter, 1997). Therefore, it was recommended that slopes >10° (17.7%) be grassed and erosion-protective crop rotations, erosion-protective tillage and fertilizer-liming treatments be used on 2–10° (3.5–17.7%) slopes (Jankauskas and Jankauskiene, 2003).

There was considerable annual soil loss variability under spring barley on the 5–10° (8.3–17.7%) slope (Figure 1.6.5). This included low values of 0.8–8.4 t ha^{-1} (1992, 1995, 1996, 1997, 1998 and 2000), moderate values of 11.6–20.6 t ha^{-1} (1984, 1987, 1988, 1990, 1993 and 1999) and high rates of 36.1–116.9 t ha^{-1} (1983, 1985, 1986, 1989, 1991 and 1994).

Annual soil losses were extremely variable during the 18-year investigation period (Figure 1.6.5). However, the correlation between total precipitation and soil loss was not significant ($r^2 = 0.21 - 0.40, p > 0.05, n = 12$). Soil erosion rates depended mostly on rainfall amount and intensity during periods when soil was unprotected by plant cover, or during snowmelt from nonfrozen slopes (Jankauskas, 1996; Jankauskas and Svedas, 2001). This accords with results from plot studies in the UK, where prolonged, low-intensity rainfall events caused relatively little erosion on bare soils and most was accomplished by short, intense (>10 mm h^{-1}) convective rainstorms (Fullen and Reed, 1986). Studies at several locations have shown that most soil erosion over an extended period occurs during a few large storms (Larson *et al.*, 1997).

Soil erosion has led to significant deterioration in the physico-chemical properties of loamy sand and clay loam Albeluvisols. Dry bulk density and percentage of clay–silt and clay fractions have increased and total porosity and water field capacity decreased. Strong acidity of E, EB and B1 soil horizons, exhumed owing to soil erosion, is a characteristic feature of eroded Albeluvisols (Jankauskas, 2000; Jankauskas and Fullen, 2002). Deterioration of soil attributes leads to decreased soil fertility (Jankauskas, 2001). The natural fertility (using barley yield as a surrogate measure) was less on eroded soils. On slopes of 2–5° (3.5–8.3%), 5–10° (8.3–17.7%) and 10–14° (17.7–24.5%) barley yield decreased by 21.7–22.1, 38.9–39.7 and 62.4%, respectively (Table 1.6.1).

1.6.4 SOIL CONSERVATION

The erosion-preventive capability of crop rotations depended on the erosion-protective properties of constituent crops and the need for these measures increases with slope gradient. The research data allowed modelling of appropriate erosion-resisting crop rotations (Table 1.6.2) and these rotations are recommended for erodible soils on 2–10° (3.5–17.7%) slopes. Long-term perennial grasses should be grown on slopes >10°

TABLE 1.6.1 Dependence of barley yield on slope steepness and soil erosion severity

Landscape segment	Severity of soil erosion	Yield[a] from 48 investigated plots		
		t ha^{-1}	Relative numbers	Decrease (t ha^{-1})
Flat land	Noneroded	18.9	100	—
Slopes of 2–5° (3.5–8.3%)	Slightly eroded	14.8	78.3	4.1
Slopes of 5–10° (8.3–17.7%)	Moderately eroded	11.4	60.3	7.5
Slopes of 10–14° (17.7–26.3%)	Severely eroded	7.1	37.6	11.8
Foot slopes	Deposited soil	19.5	103.2	—
LSD_{05}[b]		1.1		

[a]The mean of 3 years grain and straw gross yield.
[b]Least significant difference at the 95% probability level.

(17.7%). Hence sod-forming perennial grasses and erosion-protective crop rotations could assist both soil conservation and the ecological stability of the vulnerable Baltic coastal zone.

Deep soil chisel tillage can be used instead of deep mouldboard ploughing. Spraying stubble with Glifosat ($C_3H_8O_5NP$) herbicide can be used instead of the usual deep ploughing used in autumn soil tillage systems.

TABLE 1.6.2 Erosion-preventive crop rotations as soil conserving measures for fields of varying gradient

M.a.s.g.[a]	M.r.p.g.[b]	Composition of crop rotations
7–10°	80	I. 1: winter grains or spring barley; 2–5[c], perennial grasses
(11.9–17.7%)	72	II. 1: winter grains; 2, spring barley; 3–7, perennial grasses
	67	III. 1: winter grains, 2: spring barley, 3–6: perennial grasses
	63	IV. 1–2: winter grains, 3: spring barley, 4–8: perennial grasses
	63	V. 1: winter grains, 2: spring grains, 3: spring barley, 4–8: perennial grasses
	60	VI. 1: winter grains, 2: spring barley, 3–5: perennial grasses
5–7°	57	VII. 1–2: winter grains, 3: spring barley, 4–7: perennial grasses
(8.3–11.9%)	57	VIII. 1: winter grains, 2: spring grains, 3: spring barley, 4–7: perennial grasses
	50	IX. 1–2: winter grains, 3: spring barley, 4–6: perennial grasses
	50	X. 1: winter grains, 2: cereal grains with legumes, 3: spring barley, 4–6: perennial grasses
	43	XI. 1: winter grains, 2: cereal grains with legumes, 3: winter grains, 4: spring barley, 5–7: perennial grasses
	43	XII. 1: winter grains, 2: cereal grains with legumes, 3: spring grains, 4: spring barley, 5–7: perennial grasses
	40	XIII. 1: winter grains, 2: spring barley or their mixture with legumes, 3: spring barley, 4–5: perennial grasses
2–5°	38	XIV. 1: winter grains, 2: spring grains, 3: cereal grains with legumes, 4: winter grains, 5:
(3.5–8.3%)		spring barley, 6–8: perennial grasses
	38	XV. 1: winter grains, 2: spring grains, 3: cereal grains with legumes, 4: spring grains, 5: spring barley, 6–8: perennial grasses
	33	XVI. 1: winter grains, 2: spring grains, 3: cereal grains with legumes, 4: spring barley, 5–6: perennial grasses
	33	XVII. 1-2: winter grains, 3: cereal grains with legumes, 4: spring barley, 5–6: perennial grasses

[a]M.a.s.g., maximum available slope gradient.
[b]M.r.p.g., minimum requirement of grasses in a crop rotation (%).
[c]Years of crop rotations.

Soil erosion rates were reduced 2–9-fold by using these measures, while productivity remained fairly constant (Arlauskas and Feiza, 1996).

These results demonstrate the need for soil conservation measures on arable undulating environments in Lithuania. The aim of current soil erosion research is to evaluate the potential for soil conservation on eroded undulating land and to advise on policies for rural development in transitional EU Accession State economies in relation to environmental protection. Promoting soil conservation in transitional economies is crucial for effective agricultural management. In the immediate future, Lithuania could export food produce at economically competitive rates. Any such production should be provided in an environmentally friendly and sustainable way. Therefore, research data and experience of soil conservation practices on the undulating relief of the Republic are very important for sustainable agricultural development. The multi-species agro-ecosystems (sod-forming perennial grasses and grass–grain crop rotations) are potential components for both soil conservation and biodiversity strategies. It is imperative that the soil resource base is conserved for future generations. Therefore, current investigations of carbon sequestration in Lithuanian soils, funded by the Leverhulme Trust (UK), may have important benefits for environmental protection. These benefits are both national (increasing soil organic carbon and thus decreasing soil erodibility) and international (by helping to ameliorate global warming).

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1.7

Estonia

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1.7.1 PHYSICAL GEOGRAPHY

1.7.1.1 Climate

Estonia lies in the transitional zone from maritime to continental climate. In western Estonia, immediately bordering on the Baltic Sea, the climate is more maritime, whereas in the eastern part of the country, a continental climate prevails. The climate is strongly affected by cyclones developing in the north of the Atlantic Ocean. Their effect is especially strong in late autumn and early winter. The mean annual temperature is from 4.1 to 6.0 °C. The lowest mean annual temperature has been recorded at Jõgeva (1.6 °C) and the highest at Vilsandi (8.3 °C). Mean annual precipitation is 725 mm. In general, higher precipitation occurs in the uplands of central and south Estonia and the lowest in the coastal regions.

1.7.1.2 Geology

Most of the country is underlain by sedimentary rocks: Ordovician and Silurian carbonate rocks and Devonian sandstones and clays (Viiding and Raukas, 1995). These are covered by various sediments from the Quaternary. On the Ordovician and Silurian carbonate outcrops, their thickness is usually less than 5 m. Occasionally, on the so-called alvars, they are almost lacking. The Quaternary cover is at its thickest on the Haanja and Otepää Uplands (often more than 100 m). Pleistocene deposits are dominated by tills, which make up 70% of the volume and 47.7% of the area of Estonia. Glaciolacustrine and glaciofluvial deposits are also

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TABLE 1.7.1 Arable land (1990–2002)

	1990		2000		2002	
Crop	10^3 ha	%	10^3 ha	%[a]	10^3 ha	%
Cereals	397.0	35.5	329.3	29.5	259.2	23.2
Industrial crops	3.2	0.9	29.1	2.6	33.2	3.0
Potatoes	45.2	4.0	30.9	2.8	16.0	1.4
Forage crops	665.3	59.6	412.8	37.0	274.3	24.6
Total	1116.30	100.0	809.8	71.9	588.1	52.2

[a]% of the area of arable land in 1990 (1116.3×10^3 ha).

widely distributed, covering 6.8 and 3.1% of the land area, respectively (Viiding and Raukas, 1995). During the late- and postglacial period, a considerable area of Estonia was flooded by the waters of large ice-dammed lakes and the Baltic Sea (Raukas, 1992). Land began to emerge from the water as a result of the gradual uplift of the Earth's crust. This process is still in progress. At the present time, the north-western part of Estonia is rising at a rate 2.5 mm yr^{-1}.

1.7.1.3 Relief

The Estonian landscape bears distinct traces of glacial activity and is characterized by flat topography with undulating plains and small hills. The average height of the ground surface above sea level is approximately 50 m. The highest point (318.1 m) is in south-eastern Estonia.

1.7.1.4 Land Use

Land use in the 1990s was similar to that in preceding decades (Table 1.7.1). By the turn of the century, the relative share of arable land had dropped considerably whereas the share of forest land had increased. For soil erosion, arable land use is an important factor. Over time, the area of cereal crops, potatoes and flax has dropped steadily whereas the area of grassland and fallow has increased. In 2002, the area of cereal crops amounted to 44% of arable land.

1.7.2 HISTORICAL EROSION

Accelerated water erosion started simultaneously with the development of agriculture (depending on the area, as long as 5000 years ago). The process can be observed as the accumulation of slopewash sediments around the base of uplands and in wet hollows between the uplands and river beds and the occasional presence of gullies of different dimensions covered by vegetation in sloping forest areas in south-eastern parts of Estonia. Intensive wind erosion also created sand dunes in the transgression region of the Ancylos Lake (9300–7600 BP) and the Litorina Sea (7600–4000 BP).

1.7.3 CURRENT EROSION PROCESSES

Today, water erosion can be observed only on arable land (Tables 1.7.2 and 1.7.3 and Figure 1.7.1). In Estonia, soil is mostly rich in gravel and therefore relatively resistant to erosion. Soil on slopes with a gradient less than

TABLE 1.7.2 Eroded soils on arable land

Region	Arable land (10^3 ha)	Eroded soils on arable land (%)			
		Total	Slightly	Moderately	Severely
North Estonia	236.8	0.0	0.0	0.0	0.0
West Estonia	285.4	0.0	0.0	0.0	0.0
Central Estonia	366.8	2.5	1.4	0.9	0.2
South Estonia	196.8	22.9	12.7	8.5	1.7
In country as a whole	1085.8	5.3	2.9	2.0	0.4

Source: Kask (1996).

2–3° is not considered to be an erosion risk. Erosion levels are distinguished by truncated soil profiles. The assessment is based on the remaining profile: the thickness of the weakly, moderately or strongly eroded soil is <20, 20–60 and >60 cm, respectively. These soils are distributed in areas where the slope of the land is 2–5, 5–10 and >10°, respectively (Kask, 1996).

In Estonia, the degree of erosion can vary considerably from year to year. For example, on 9 September 1979, heavy rain (ca 100 mm) caused a loss of soil in a field with a new growth of rye to an extent that exceeded the total erosion that had taken place over the past centuries (the estimate was based on fresh slopewash sedimentation on old layers of sediments in a closed hollow). On slopes with a gradient of 5–10° one can often find places where the volume of rills is 25–50 and less frequently 50–100 $m^3\ ha^{-1}$ and in extreme cases this may also be considerably larger. The volume of the largest known scour gully forming in a field as a result of one downpour (on 9, September 1979) was assessed to be 67 m^3.

Over time, the rills not levelled in the course of cultivation become gullies. The Otepää and Haanja Uplands are remarkably hilly, with relative heights up to 70 m. The slopes of the hills are steep, mostly 5–20° and in rare cases even up to 30°. Soil erosion is intensive on agricultural lands. On the Haanja Upland, in an area with rough terrain, 1.7 gullies per square kilometre were counted (Kask, 1957). The number of gullies is higher on the edges of old valleys. On the edges of Kütioru valley, 11 gullies were counted along a 2.3-km stretch. The length, depth and width of the largest gullies were 380, 42 and 100 m, respectively (Heinsalu, 1988).

Until 1950, horses were mostly used to work the fields. Slopes of up to 30° were cultivated. Following the transition to mechanized land cultivation, seriously eroded fields with gradients exceeding 8–10° were soon left out of the production cycle. Therefore, the area of land with accelerated erosion started to fall. In 1988, the area of strongly eroded land amounted to 57/700 ha, which is approximately 70% of the former eroded area (Kask, 1996). Today, the area of land with accelerated erosion has dropped even more. Uncultivated arable land is used as grassland or is waiting for afforestation.

TABLE 1.7.3 Soils at risk from wind erosion on arable land

Region	Arable land (10^3 ha)	Total		Slightly		Moderately	
		10^3 ha	%	10^3 ha	%	10^3 ha	%
North Estonia	236.8	51.6	21.8	5.9	2.5	45.7	19.3
West Estonia	285.4	66.2	23.2	5.5	1.9	60.7	21.3
Central Estonia	366.8	47.8	13.0	2.7	0.7	45.3	12.3
South Estonia	196.8	35.5	18.0	1.0	0.5	34.5	17.5
In the country as a whole	1085.8	201.1	18.5	15.1	1.4	186.2	17.1

Source: Kask (1996).

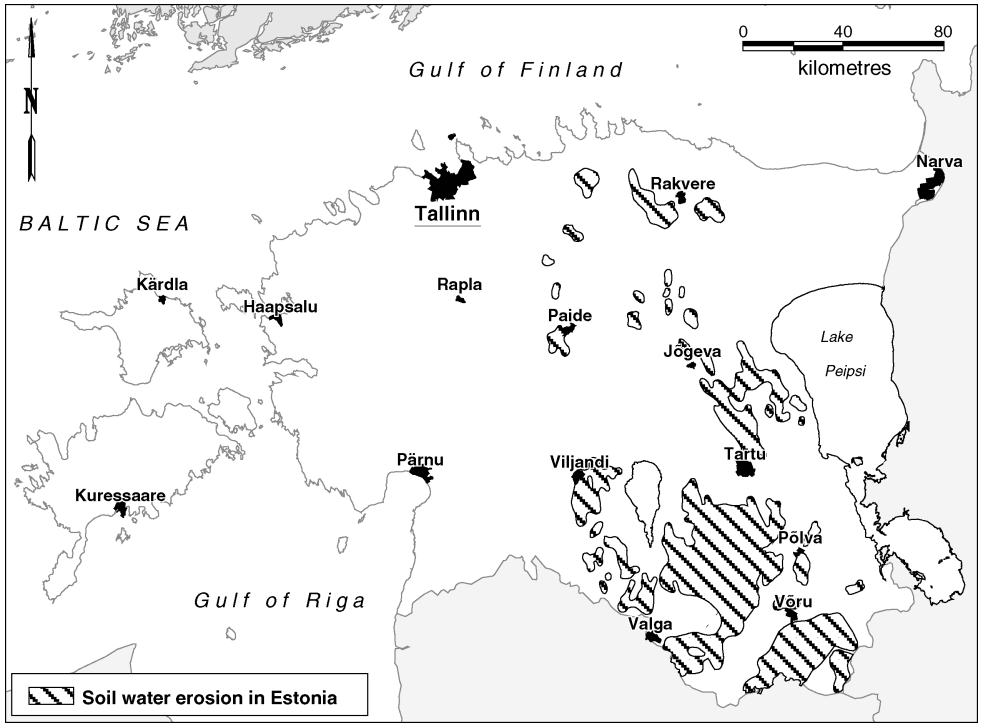

Figure 1.7.1 Soil water erosion in Estonia

The working capacity of modern soil cultivation machinery and implements, and therefore also their increased speed in the working process, have contributed to increased mechanical transportation of soil down the slope (tillage erosion).

1.7.4 MAJOR ON- AND OFF-SITE PROBLEMS AND RELATED COSTS

In Estonia, when eroded areas with uneven terrain are being mapped (1:5000 and 1:10 000), a distinction is made between eroded (off-site) and slopewash (on-site) soils. The ratio of such soils in the area suffering most strongly from erosion, the Otepää and Haanja Uplands, is in the range 3–4:1. The humus and nitrogen contents of off-site soil are 0.3–0.95 of that characterizing on-site soil. The contents of phosphorus and potassium in the ploughed layer of the soil depend on soil type and level of erosion. Where the ploughed layer of the soil includes some material from the elluvial horizon below (A2e from E horizon), the content of the aforementioned elements in the ploughed layer decreases. Where the ploughed layer includes some material from the illuvial horizon (B and BC horizon), the content in the ploughed layer will exceed their content in on-site soil. As the erosion process advances, the acidity of off-site soils drops. This is particularly noticeable in off-site soils that are high in residual carbonates; the share of such soils among the eroded soils of Estonia is relatively high.

The concentrations of humus and nutrients in cumulative (on site) soil are lower than those in buried and off-site soil. Thanks to the considerable thickness of the cumulative humus horizon (up to 1 m), such soil, as a rule, has high fertility potential.

Productivity of off-site soil depends on the level of erosion as follows: in weakly, moderately and strongly eroded soil, 0.85, 0.70 and 0.50, respectively, of the respective indicator for on-site soil. Productivity of cumulative soils depends on the thickness of deposited layers and can amount to 1.05–1.30 of the respective indicator of buried soil. Yields of agricultural crops obtained from eroded and cumulative soil can differ as much as 10-fold, even when cultivated within the same field.

1.7.5 SOIL CONSERVATION AND POLICIES TO COMBAT EROSION AND OFF-SITE PROBLEMS

In Estonia as yet, no special measures have been taken to combat soil erosion. Over time, it became customary to cease using a field when it was no longer suitable for cultivation, and forest recolonized it. Decreased fertility of the soil, caused by erosion, was not the only reason to cease cultivation of former fields – factors such as the unsuitability of small, steeply sloping fields for mechanized cultivation also played a part (Kask, 1964). During the second half of the 20th century, reorganization of land tenure was begun to make more rational use of the resources available. The need to combat soil erosion was also considered in the process. A large share of ancient fields were left fallow, overgrown with forest or were afforested; new (irrigated) grasslands were established and the share of different varieties of grass increased in rotation schemes. In 1970s, complex land amelioration was started in some hilly regions and more successful large-scale farms. Related activities included the levelling of micro-relief, restoration of fertility of eroded soils, establishment of reservoirs in wet hollows, construction of irrigation systems and renovation or construction of roads. The amelioration efforts were soon dropped owing to their high cost and changing market conditions after Estonia regained its independence. In 2004, the Estonian Agri-Environmental programme was started with several measures which should mitigate the problem of wind and water erosion of soils.

In Estonia, no restrictions at national level have been imposed on the use of land at risk of erosion; there are also no mandatory requirements intended to slow erosion and restore the fertility of soil. There have been numerous articles, manuals, seminars, etc., giving recommendations by scientists and describing good examples (Penu, 2005). As much as possible, agricultural production processes consider these recommendations. When considered as a whole complex of problems, insufficient attention is being paid to soil erosion in Estonia; this applies both to government agencies and to practical production activities.

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1.8

European Russia and Byelorus

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1.8.1 INTRODUCTION

The plains and uplands of the European part of the Russian Federation (Russia) and the Republic of Byelorus (Byelorus), with a total area of 4.03 $(3.82 + 0.21) \times 10^6$ km^2 are surrounded by the Ural Mountains in the east, the Barents Sea in the north, Finland, the Baltic States, Poland and the Ukraine in the west and the Azov and Black Sea, Caucasian Mountains and Caspian Sea in the south (Figure 1.8.1). The processes of erosion and sedimentation are most clearly manifested in (1) sheet and rill erosion on slopes, (2) gully erosion and (3) deposition of sediments in dry valleys and river systems. These processes are controlled by topography, soil erodibility, melt water and rainfall erosivity, vegetation cover and land use. The combination of land-use history and variations in the above biophysical factors produced a history and pattern of erosion that is unique to this area. In this pattern, the influence of geographical zoning is clearly evident, and is expressed in changes of the climatic and landscape conditions over the territory, in the latitudinal extent of vegetation and soil zones and in socio-economic conditions. The development of intensive agriculture, beginning in the 15–16th centuries, first occurred in the forest zone, then in the forest–steppe and subsequently the steppe zone.

1.8.1.1 Landforms

Three main latitudinal belts with different terrain types are characteristic of the territory. The northern belt of fresh glacial and fluvioglacial relief occupies the northern megaslope of the Russian Plain (Onega, Severnaya Dvina, Mezen' and Pechora River basins) and the Upper Volga basin. Here, narrow chains of uplands separate

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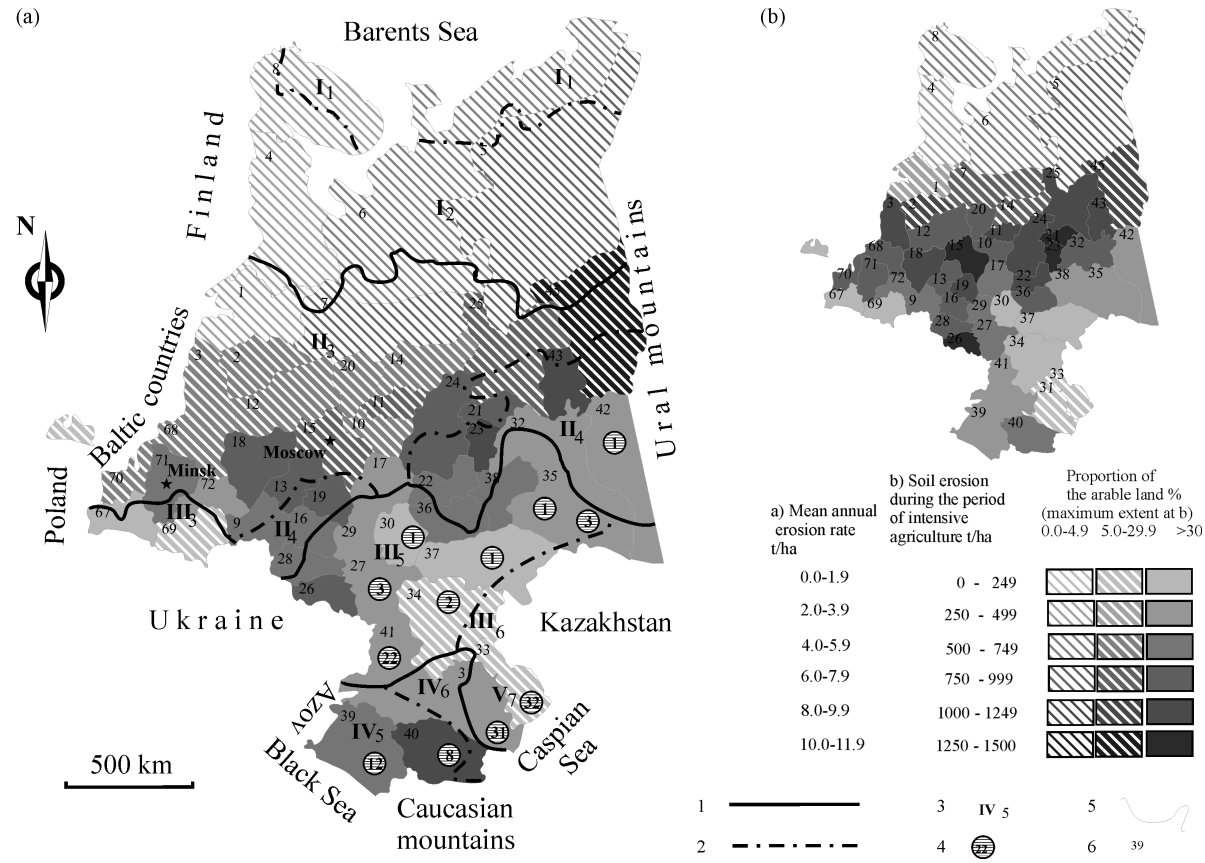

Figure 1.8.1 (a) Contemporary (calculated) soil erosion rates in European Russia and Byelorus and (b) soil loss (calculated) during the period of intensive agriculture. Key: 1, boundaries between natural erosion zones; 2, boundaries between main regions of man-induced erosion; 3, zone and region indices: (I) melt-water erosion; (II) melt-water and rainfall erosion; (III) mainly rainfall erosion; (IV) rainfall erosion without snow melt; (V) occasional erosion; and (1) reindeer breeding; (2) sporadic farming; (3) mixed farming – cultivation and stock-raising, with highly selective land use; (4) intensive tillage with low selectivity; (5) land fully exploited for cultivation; (6) tillage and grazing; (7) grazing and sporadic cultivation; 4, percentage of district area affected by wind erosion; 5, administrative district boundaries; 6, district indices as in Table 1.8.1

broad lowlands. Owing to the deep seasonal soil freezing and generally high soil moisture content, arable lands are mainly situated on the steeper drained slopes with mean lengths of 130–380 m and inclination of 2–5° (up to 9–10°). The middle belt of the old glacial and fluvial relief consists of a sequence of undulating lowlands and uplands, from the Poles'ye and Pridneprovskaya lowlands in the west to the Middle Russian upland and Oksko-Donskaya lowland in the centre and the Privolzgskaya upland and Zavolzhskaya lowland in the east. Here agricultural selectivity of relief is less marked: only the steepest slopes are not ploughed. Therefore, the difference between arable fields in the lowlands (inclination ~1–2°, slope length 200–300 m) and in the uplands (inclination 4–8°, length ~400 m) is pronounced. The southern belt of fluvial and coastal relief has a similar structure and consists of the Asov-Kuban' lowland in the west and Prikaspiyskaya lowland in the east, separated by the Stavropol' upland. Here the slope inclination of arable land is extremely varied at 0.5° in the lowlands and 5° in the uplands, but the slope length is more uniform: 600–650 m. All these morphological

units (and their smaller elements) are characterized by typical probability density functions and mean values of the Universal Soil Loss Equation (USLE) *LS* factor: in uplands it ranges from 1.5 to 2.5 and up to 3, in lowlands it usually ranges from 0.4 to 0.75, and the lowest value is 0.25 (Litvin *et al.*, 2003).

1.8.1.2 Soil Erodibility

European Russia is the classical area for the latitudinal extent of soil zones, first discovered by Dokuchaev (1883). The northernmost is the zone of tundra gley and gley–illuvial soils, which grade to Podzols under the coniferous forests of the northern and middle taiga and Sod-Podzols of the southern taiga. Further south, the zone of grey forest soils was formed under broad-leaved forests and a broad zone of Chernozems corresponds to the forest–steppe and typical steppe. In the dry steppe, dark-brown (Chestnut) soils are predominant. Grey–brown and light-grey–brown soils occupy the southernmost desert zone.

Soils differ in their susceptibility to erosion, determined by their mechanical composition, organic matter content, structure and rate of formation. A commonly used index of erodibility is the USLE *K* factor. Resistance to erosion increases from north to south from Podzols to grey forest soils and Chernozems, and then decreases in the dark-brown soils and desert and semi-desert soils. Well-structured Chernozems and dark-grey forest soils with a high organic matter content and loamy texture are most resistant (*K* as low as 0.11–0.16 t ha^{-1} per erosivity unit), the least resistant being Podzols, Sod-Podzols, desert grey–brown and light-grey–brown soils (*K* reaching 0.46–0.53 t ha^{-1} per erosivity unit). The same trend was found for the formation rate of humus (A) horizons: it is 0.1–0.2 mm yr^{-1} for Podzols, 0.2–0.3 mm yr^{-1} for Sod-Podzols, 0.35–0.4 mm yr^{-1} for grey forest soils, 0.4–0.45 mm yr^{-1} for Chernozems, 0.2–0.3 mm yr^{-1} for dark-brown soils and 0.1 mm yr^{-1} for light-brown and solodic soils (Gennadiev *et al.*, 1987).

1.8.1.3 Climatic Factors Affecting Erosivity

The climate is temperate–continental with a long, severe winter and short summer. The main climatic factors influencing water erosion are snowmelt runoff and rainfall. The period of snowfall extends from mid-October until early May in the north and from late December until late February in the south. The depth of water flow during the snowmelt period is determined by the amount of water in the snow at the start of the melt and by the runoff coefficient. The late-winter water content of snow is greatest in north-eastern European Russia, decreasing towards the south and west. In the south, snow cover is absent in some years. The value of the runoff coefficient in the thaw period depends on soil saturation and the extent of soil freezing. High runoff coefficient values in the northern, north-western and central regions can be explained by the soils being moist in autumn and deeply frozen in winter. The decrease in the coefficient eastwards is the result of lower early winter soil moisture contents, despite the extent of freezing. Towards the south there is a decrease in both the soil moisture content and the degree of freezing. Owing to the similar spatial distribution of the main factors determining runoff during the melt, runoff in European Russia decreases rapidly from north to south (from 200–220 to 10–20 mm) and from the central regions to the east and west.

Runoff during the period of summer rains is determined by the amount of rainfall and the runoff coefficient. The value of the runoff coefficient depends on slope morphology, vegetation cover and soil infiltration capacity, varying within broad limits over the territory. Rainfall energy and its erosive capacity, expressed by the rain erosivity (*R*) of the USLE, are closely correlated with amount of rainfall. The distribution of rainfall, and that of *R*, is variable over European Russia, but it has a tendency to increase from north to south and from east to west. The proportion of rainfall in total precipitation is ~50–70% in the north and up to 90% in the south of the territory. The proportion of melt water in total runoff is much greater than that of the rain water, because runoff coefficients during the snow thaw period are higher than in the rest of the year.

1.8.1.4 Vegetation Cover

In its natural state, the vegetation cover of European Russia and Byelorus was in all areas dense enough for erosion to be slow. Under present conditions in the northern part of the territory, where the natural plant cover of tundra and taiga is mostly undisturbed, erosion rates remain very low. In the agricultural areas, vegetation cover is almost entirely determined by land use. Similarities of crop rotation and cultivation systems in various zones have substantially reduced the regional variability of this changeable factor. In European Russia and Byelorus as a whole, the protective role of vegetation decreases towards the south and south-west, with a diminishing proportion of perennial grasses in the crop-rotation system and a higher proportion of repeated sowing of inter-tilled crops. In the taiga zone, crop vegetation cover in the fields reduces erosion by 40–70% during the spring snow melt and by 75–85% during summer rains. In the mixed and deciduous forest zone, this reduction is 20–60 and 70–75% and in the steppe 15–20 and 60–70%, respectively.

1.8.1.5 Land Use

Agriculture became a permanent part of the economy of the Eastern Slavs towards the late 15th century, as the Muscovite State gained control of most of European Russia. Clearing of forests in the southern half of the forest zone then took place. In the 16th century, new territories were opened up and settlement established in the central Chernozem, central Volga and central pre-Ural regions. An intensive agriculture developed, with a fallow system in the steppe region, and clearing–burning and fallow systems in the forest–steppe and forest zones (Krokhalev, 1960). At the beginning of the 18th century, the area of arable land increased rapidly. A three-field system (winter wheat, summer crops and fallow) began to be used in the central regions of European Russia and the area of industrial crops (such as flax) began to increase, although it still remained very small. The most favourable arable land was largely found on the southern slopes of morainic hills with gradients of 2–4° directly adjoining river valleys, along which most settlement developed. Ploughing was restricted to the hillslopes. As a result, the length of the fields did not exceed 150–220 m. At the end of the 18th century, the settlement of the southern and south-eastern parts of the territory began. As people moved southwards into a region with greater local relief, they began to cultivate slightly longer and steeper fields: slopes of 5–7° were cultivated, often 300–400 m long. Ploughing along (up and down) the slopes was retained, as in the forest zone, and promoted gully formation (Sobolev, 1948).

Reliable agricultural data for Russia were obtained during a General Survey in the late 18th century (Tsvetkov, 1957). This period saw a gradual decrease in arable fertility as increasing production of cereals for export displaced cattle rearing. The three-field system of rotation was at this time applied over most of the territory. In the first half of the 19th century, different agricultural systems began to be used. In the Yaroslavl' and Moscow districts, for example, a four-field crop rotation system (fallow, winter wheat, clover, and summer crops) was introduced beginning in the 1820s. A crop-rotation system without fallow was used in the western regions (Byelorus). Most landowners, however, retained the traditional three-field system. Commercial cattle rearing was predominantly retained in the south and southeast.

After the abolition of serfdom in 1861, radical changes occurred in the agriculture of Russia. There was a marked increase in crop specialization, and only the north-east retained the clearing–burning system for cereals. Intensive ploughing began in the south-east and south in the Stavropol' steppes, with the fallow system retained. Flax was now sown over a wide region in the north-west and Upper Volga region as far as Nizhniy Novgorod, being incorporated in the multi-field rotation (fallow–rye–oats–2 year grass–flax–oats). In the rest of the territory, outside the Chernozem zone, eight-field rotations were used, in which cereals alternated with fallow, grass and potatoes. Western regions now began to specialize in beet production, which was included in a 10-field rotation or in an improved cereal rotation (fallow–winter cereals–beet–summer cereals). The ploughed area in southern forest and forest–steppe zones of European Russia reached its maximum in late 19th century (Table 1.8.1). In

TABLE 1.8.1 The main characteristics of erosion in European Russia and Byelorus. Columns: 1, country; 2, district index; 3, district name; 4, district area (10^3 ha); 5, maximum proportion of arable land (%)/year when this maximum occurred; 6, mean annual rate of sheet and rill erosion on arable land in the 1970–80s (t ha^{-1}) (calculated); 7, amount of sheet and rill erosion during the period of intensive agriculture (10^6 t) (calculated); 8, volume of gullies >70 m long (10^6 m^3); 9, area, affected by wind erosion (10^3 ha) [a value of 0 means small (<1000 ha) extent of wind erosion]

1	2	3	4	5	6	7	8	9
Russian Federation	1	Leningradskaya	8531	16.3/1868	2.6	683.6	1.03	0
	2	Novgorodskaya	5447	12.4/1868	4.5	734.8	1.47	0
	3	Pskovskaya	5540	32.3/1868	5.8	1822.5	1.46	0
	4	Kareliya	18052	2.5/1887	2.6	167.2	0.00	0
	5	Komi	43371	1.2/1887	6.9	499.3	1.27	~10
	6	Arkhangelskaya	57127	0.6/1950	4.9	166.1	2.83	~10
	7	Vologodskaya	14451	6.2/1950	6.1	802.0	1.02	0
	8	Murmanskaya	14493	0.1/1796	2.6	7.6	1.58	0
	9	Bryanskaya	3485	57.5/1887	4.1	1077.4	14.84	1.2
	10	Vladimirskaya	2912	43.8/1868	5.5	1134.7	10.92	0
	11	Ivanovskaya	2342	43.8/1868	6.5	1218.9	3.59	0
	12	Tverskaya	6020	31.7/1868	5.3	1554.9	2.56	0
	13	Kaluzhskaya	2978	53.7/1868	7.4	1589.2	12.79	0
	14	Kostromskaya	6020	20.6/1868	5.6	1128.4	7.29	0
	15	Moskovskaya	4689	39.0/1861	7.7	2413.7	8.85	0
	16	Orlovskaya	2465	68.2/1980	5.3	1349.8	35.90	0
	17	Ryazanskaya	3961	56.0/1868	3.5	1344.1	11.56	20.5
	18	Smolenskaya	4978	38.1/1868	7.7	2120.5	13.04	0
	19	Tul'skaya	2568	74.0/1887	7.5	2324.8	15.19	0
	20	Yaroslavskaya	3620	35.1/1868	5.4	1206.3	3.81	0
	21	Mari-El	2237	49.6/1887	7.1	1678.6	12.56	0
	22	Mordoviya	2613	62.4/1887	6.0	1928.1	41.22	0
	23	Chuvashiya	1835	49.6/1887	8.6	1808.4	24.14	0
	24	Nizhegorodskaya	7462	42.5/1887	6.7	3913.8	13.28	0
	25	Vyatskaya	12035	34.1/1887	6.2	4092.3	12.04	0
	26	Belgorodskaya	2713	72.0/1887	7.8	2433.0	43.09	0
	27	Voronezhskaya	5222	69.7/1887	3.6	1907.4	33.11	163.8
	28	Kurskaya	3000	72.8/1887	6.0	1991.2	19.47	0
	29	Lipetskaya	2405	70.0/1950	9.2	914.8	46.82	0
	30	Tambovskaya	3446	66.5/1980	1.7	685.5	14.39	34.4
	31	Kalmykiya	6855	13.9/1980	2.3	196.5	6.01	2103.2
	32	Tatarstan	6784	55.4/1980	2.9	3227.1	28.05	0
	33	Astrakhanskaya	5303	8.0/1980	0.3	10.1	1.58	1692.8
	34	Volgogradskaya	11294	51.7/1980	1.7	822.5	32.67	234.8
	35	Samarskaya	5360	57.8/1980	2.3	950.9	25.22	60.2
	36	Penzenskaya	4335	62.4/1887	4.3	2661.3	32.90	0
	37	Saratovskaya	10124	63.1/1980	1.9	1473.7	28.56	124.6
	38	Ul'yanovskaya	3718	53.3/1887	4.4	931.9	3.52	10.6
	39	Krasnodarskiy	8328	58.4/1950	5.4	1780.9	8.71	1023.0
	40	Stavropol'skiy	7279	66.1/1950	10.0	3346.3	14.69	617.8
	41	Rostovskaya	10097	60.8/1980	3.1	1767.7	18.21	2227.0
	42	Bashkiriya	14294	35.3/1980	3.0	1621.2	1.38	143.0
	43	Udmurtiya	4206	36.7/1980	9.7	1829.6	25.08	30.0

(Continued)

TABLE 1.8.1 (*Continued*)

1	2	3	4	5	6	7	8	9
	44	Orenburgskaya	12369	36.5/1980	2.1	1156.8	1.62	384.0
	45	Permskaya	16024	16.4/1980	12.1	3135.2	8.09	0
Byelorus	67	Brestskaya	3278	41.3/1868	1.2	275.7	1.19	0
	68	Vitebskaya	4005	45.2/1868	4.5	1403.1	2.42	0
	69	Gomel'skaya	4036	32.7/1796	0.9	180.2	2.13	0
	70	Grodnenskaya	2511	41.3/1868	4.9	1055.0	3.50	0
	71	Minskaya	4021	32.7/1796	4.5	993.1	1.62	0
	72	Mogilyevskaya	2908	45.7/1868	3.1	687.1	5.54	0

the grain-producing areas of the Central Chernozem zone, the crop rotation was often broken and grains sown in three or four consecutive years. It was also a period of increase in the numbers of land users who owned small fields: 60% of peasants owned land with an area of <10 ha. At this time, in both the forest and forest–steppe zones steep slopes of dry valleys, unsuitable for cultivation, were ploughed. Narrow strips along the slope represented the plots of land. These strips were separated from each other by deep plough lines, which concentrated flow and promoted gully formation. The length of the ploughed parts of slopes did not exceed 100–150 m in the forest zone, 200–250 m in the forest–steppe and 300–350 m in the steppe.

The area of arable land was reduced during World War I, followed by a period of significant private involvement in agriculture during the 1920s. This period ended with general collectivization beginning in 1928. Crop rotations changed to multi-field, somewhat improving soil protection against erosion by increasing vegetation cover. The area of cereal crops decreased from 80–85 to 70–75%, as industrial (mainly sunflower and sugar beet) and fodder crops increased. Field sizes increased because the area of fallow land was reduced and tractors were introduced. Development began in the virgin lands of the lower Volga, in the pre-Urals, the pre-Caucasus and the lower Don River basin.

During World War II, the area under crops was again everywhere reduced, by a factor of not less than three. By the late 1950s, the area of crops had been restored, owing to the use of tractors, combine harvesters and other techniques. A change in the structural and hydrological properties of soils began at this time, resulting particularly from the increased loading by machines, and causing increased runoff and erosion. After the 1950s, all arable land in the steppe zone of the territory was used, with the last increase in ploughed area coming about by cultivating floodplains, which had previously been used for pastures. The near doubling of the weight and size of tractors continued the process of making tilled soils more susceptible to erosion. Some reduction in the area of ploughed land in the forest zone and forest–steppe zone occurred in the 20th century, as the most eroded areas were excluded from cultivation and some lands were used for urban development and mining.

The 1970–80s were characterized by year-to-year variations of only 1–2% in the area of cultivation. Disc ploughing of 10–15% of the Chernozem zone increased the resistance of these soils to erosion. Outside this zone, the extensive use of grain–fodder systems with 30–40% perennial grasses in the rotation of these crops also increased resistance to erosion by increasing vegetation cover.

1.8.2 SPATIAL DISTRIBUTION OF SHEET AND RILL EROSION

1.8.2.1 Contemporary Processes

The spatial distribution of soil loss in an area with such diverse climate, soil and relief as European Russia and Byelorus is extremely complicated (Litvin *et al.*, 2003). Substantial changes in the climatic parameters of the

area, such as precipitation and the proportion of rain in relation to snow, produce various zonal combinations of fundamentally different forms of erosion: melt-water erosion and rainfall erosion (Figure 1.8.1). In the north lies zone I of melt-water erosion and further south zone II of melt-water and rainfall erosion. At its northern limit, the severity of soil loss from both types of erosion is approximately equal. At the southern limit of zone II, the rate of melt water erosion is roughly equal to the rate of natural soil formation. The northern limit of zone III, in which rainfall erosion predominates, corresponds to the limit of the area with irregular snow cover. Further south, in zone IV, melt water erosion rarely occurs, and the proportion of rainfall erosion is much higher. The southernmost zone V of occasional rainfall erosion is a region where erosion by water is very rare and extremely short-lived.

Agricultural land use represents another basis for zonation of the erosion status. The distribution and extent of agriculture, the proportion of tillage and the relation between pasture and arable land determine erosion severity. For example, in region 1 with its reindeer pastures, water erosion occurs only within highly disturbed oil and gas fields, and the pasture itself is subjected mainly to wind erosion if overgrazed. In patchy farming region 2, erosion severity on cultivated slopes is substantial, but the total soil loss is small because arable land comprises only a few percent of an area, which is mostly forest or tundra. Soils in the north generally receive excessive moisture. Therefore patches of well-drained land on rather steep slopes are cultivated first, while flat interfluves remain forested or swampy.

In northern region 3 of mixed farming–cultivation and stock raising – also with highly selective land use due to a high spatial variability of the landscape, the distribution of arable land and pasture is complicated, but the rate of erosion on the arable land is fairly constant owing to the similarity of terrain selected for farming. In regions 4 and 5 of intensive and maximum extent of agriculture, arable land comprises up to 60–70% of the area and the erosion rate is both high and variable. In region 6, the pasture area increases and mixed farming (cultivation and stock raising) prevail again. In region 7, sheep grazing is the main type of agriculture.

Local events of intensive runoff cause close to catastrophic erosion rates. Khokh and Zhilko (1981) reported an erosion rate of 46 t ha^{-1} on Sod-Podzols in Byelorus during the snowmelt spring period in 1972. Medvedev and Shabaev (1991) measured an erosion rate of 53.5 t ha^{-1} during spring 1974 on the Privolzhskaya upland, when rainfall combined with melt-water runoff. The same situation on the Azov Sea coastal plain caused an erosion rate of 25 t ha^{-1} for one event (Poluektov, 1984). Catastrophic summer rainfall (72 mm on 23 May 1967) caused an erosion rate of 220 t ha^{-1} from a potato field and 84 t ha^{-1} from a rye field in Byelorus (Zhilko, 1976). On 20–21 August 1976, 192 mm of rainfall caused the formation of ephemeral gullies 200 m long, 2 m wide and 0.2–0.3 m deep and a soil loss about 50–100 t ha^{-1} in an area of 2000 ha in the Kursk district (Gerasimenko and Rozhkov, 1976). About 55 mm of rainfall in the Tula district during 2 h on 10 August 1997 brought about a soil loss of 22–59 t ha^{-1} (Golosov *et al.*, 1999). Such runoff and rainfall events with a 10–20-year return period produce 70–80% of the total long-term sheet and rill erosion.

The long-term erosion from large territories was calculated. The Universal Soil Loss Equation (Wischmeier and Smith, 1978) was used to calculate soil loss from rainfall. Soil loss during snowmelt was calculated using the model of the Russian State Hydrological Institute (Anon., 1979). The models were modified for European Russia conditions (Larionov, 1993), verified with measurements and showed good results (Litvin *et al.*, 2003). A schematic map (Figure 1.8.1a, Table 1.8.1, column 6) shows the average calculated severity of sheet and rill erosion, specified for administrative districts. On the Baltic seaboard the average soil loss from the arable land on major uplands is 5–7 t ha^{-1} yr^{-1} (in the south, 8–9 t ha^{-1} yr^{-1}) and on the lowlands 1.0–1.5 t ha^{-1} yr^{-1}. On glacial landforms in the uplands it reaches 10–12 t ha^{-1} yr^{-1} and on glacial-lake and fluvioglacial plains $\sim$2 t ha^{-1} yr^{-1}. Similar relationships are found between soil loss from uplands and plains in central European Russia: Middle Russian Uplands, 7–8 t ha^{-1} yr^{-1}; Dnieper Valley, 12–14 t ha^{-1} yr^{-1}; and the Oka-Don and Dnieper lowlands, 0.5–2.0 t ha^{-1} yr^{-1}. By contrast, the lowest erosion rate in the middle of the Pripyat', wooded lowland in Byelorus, is $<$0.5 t ha^{-1} yr^{-1}. The southern Stavropol', upland stands out as having the highest soil loss: 15–20 t ha^{-1} yr^{-1}. The lowlands are characterized by low rates of soil loss: the

Caspian Plain loses $<0.5\ t\ ha^{-1}\ yr^{-1}$ (this area is hardly subject to erosion at all) and the central Black Sea Plain $2\ t\ ha^{-1}\ yr^{-1}$.

The mean calculated rate of sheet erosion on arable lands is $4.8\ t\ ha^{-1}\ yr^{-1}$ in European Russia. On 13.2% of the arable land the rate of erosion is $<0.5\ t\ ha^{-1}\ yr^{-1}$, on 33.6% it is within the range 0.5–2.0 $t\ ha^{-1}\ yr^{-1}$, on 26.8% 2.0–5.0 $t\ ha^{-1}\ yr^{-1}$, on 13.3% 5.0–10.0 $t\ ha^{-1}\ yr^{-1}$, on 9.1% 10.0–20.0 $t\ ha^{-1}\ yr^{-1}$ and on 4.0% the erosion rate exceeds $20.0\ t\ ha^{-1}\ yr^{-1}$. In Byelorus the mean rate of erosion is lower: $3.6\ t\ ha^{-1}\ yr^{-1}$. The distribution of different levels of erosion on the arable land is 60.5, 3.6, 11.7, 9.8, 11.1 and 3.3%, respectively, with significant bias to low rates of soil loss. The calculated annual soil loss in European Russia in the 1970–80s was $\sim 420 \times 10^6$ t from 88.7×10^6 ha of arable land. In Byelorus the total soil loss was $\sim 20 \times 10^6$ t from 5.8×10^6 ha of arable land.

1.8.2.2 Historical Sheet and Rill Erosion During the Period of Intensive Agriculture

Change in the degree of erosion in European Russia and Byelorus may be calculated using recent rates of slope erosion and estimates of change in the principal factors causing erosion: the area under cultivation, precipitation and land use. Allowing for the relative change in the values of erosion factors, retrospective calculations were made to estimate the intensity of erosion (Sidorchuk and Golosov, 2003). The volume and the rate of soil loss for the period of intensive agriculture were thus calculated (Figure 1.8.1b, Table 1.8.1, column 7).

According to those estimates for the period from the 18th to the 20th century, erosion was related to the spatial differentiation of erosion factors and the history of the spread of cultivation in European Russia and Byelorus. In the 18th century, erosion was highest in the most densely populated and cultivated area of the Sod-Podzols. Two main areas stand out as having the most intense erosion: in the west, the Smolensk–Moscow region, and in the east, the middle Volga valley. On 94% of arable land (88% in Byelorus) the eroded layer did not exceed 10 cm. In the Smolensk–Moscow region and the middle Volga valley the eroded layer reached 20–30 cm on 8–9% of arable land. The depth of erosion was up to 20 cm on 12% of arable land in Byelorus (25–40% in Brestskaya and Vitebskaya districts). However, for Sod-Podzol soils, where the humus horizon does not exceed 15–20 cm and the rate of soil formation is no more than 2–3 cm in 100 years (under natural vegetation), such erosion rates are sufficient to produce moderate to severely eroded soil.

In the 19th century, the heaviest erosion still occurred in the long-tilled areas of the Sod-Podzols. Erosion increased after the reform of 1861 as a result of the ploughing of both land previously deemed unsuitable for cultivation and steeper hillsides. Consequently, by 1887 in the Moscow area of heavy erosion, the eroded layer exceeded 10 cm on 40% of arable land and 30 cm on 22% of arable land. In the middle Volga valley, on 63% of arable land, erosion reached >10 cm, and on 14% it was >30 cm. In Byelorus, where the arable land area was more stable and even decreased in several districts, the depth of erosion exceeded 20 cm on only 7% of arable land.

The beginning of land tillage in the Chernozem (black-earth) forest–steppe and steppe belt of European Russia led to the formation of the south-western and central black earth zones of intensive erosion. In the south of the Belgorod district the eroded layer was >10 cm deep on 30% of arable land. However, for the developed Chernozems, which typically have a humus (A) horizon up to 80–90 cm thick, and a soil formation rate under natural vegetation of 4–4.5 cm per 100 years, such erosion rates led to changes in soil structure, which did not exceed the range of natural variation. Therefore, they were not always recorded in soil erosion surveys.

In the 20th century (for our calculations, 1887–1980), the intensity of erosion on long cultivated land on the Sod-Podzol soils decreased substantially. This was connected with a reduction in the tilled area, mainly because ploughing ceased on the most heavily eroded land and on steep slopes. This accounts for the fact that the total erosion of plough-land increased only slightly. In the Central Chernozem Belt, erosion to a depth of >30 cm covered 7% of arable land in the Belgorod district and up to 22% in the Tula district. A southern erosion area developed on newly cultivated land in the Stavropol district.

TABLE 1.8.2 Calculated sheet and rill erosion (10^9 t) during the period of intensive agriculture

	Period						
Country	1950–80	1887–1950	1868–87	1861–68	1796–1861	1763–96	1696–1763
European Russia	12.65	21.1	16.22	1.68	3.54	3.83	2.11
Byelorus	0.67	0.84	1.52	0.16	0.34	0.46	0.28

Calculations show (Sidorchuk and Golosov, 2003) that in European Russia, during the period 1696–1796, a total of 5.9×10^9 m^3 of soil was washed away by sheet and rill erosion, in 1796–1887 30.8×10^9 m^3 and in 1887–1980 33.8×10^9 m^3. The constant increase in the volume of soil loss per unit time (Table 1.8.2) is due to an increase in the area under cultivation. Soils in the Sod-Podzol area are the most affected, particularly in the Middle Russian and Volga uplands, in the north and south-west of the Central Chernozem Belt (Figure 1.8.1b). The total volume of calculated soil loss from slopes in European Russia over the period from the 18th to the 20th century inclusive amounts to 70.5×10^9 m^3.

In Byelorus, during the period 1696–1796, 0.74×10^9 m^3 of soil were washed away by sheet and rill erosion, in 1796–1887 2.02×10^9 m^3 and in 1887–1980 1.51×10^9 m^3. The maximum volume of soil loss in the 19th century is due to the maximum area of arable land. Soils in the Vitebskaya and Grodnenskaya districts in north-west Byelorus were the most affected (Figure 1.8.1b). The total volume of calculated soil loss from slopes in Byelorus over the period from the 18th to the 20th century inclusive amounts to 4.3×10^9 m^3.

This huge amount of eroded soil resulted in a substantial reduction in soil depth, mainly in humus and illuvial horizons ($A + B_1$). On the morainic hills of the Valday Experimental Station in the Novgorod district, the cover layer of silt deposits with Sod-Podzol soil is 25–38 cm thick under the forest. This depth was used as the reference depth of noneroded or slightly eroded soil. Under the arable land, the silt deposits were 3–14 cm deep and in 30% of the area they were completely washed away (Lidov, 1976). In the Ul'yanovsk district the depth of $A + B_1$ horizon of noneroded Chernozems is 80–90 cm on flat land and 55–60 cm on gentle slopes. The mean thickness of these horizons for the complicated sporadic pattern of slightly eroded and moderately eroded soils on the slopes between ephemeral gullies is 30–40 cm. This thickness decreases to 10–20 cm in ephemeral gullies with a density of $\sim$3 km km^{-2} (Lidov *et al.*, 1973). At the Ergeni upland in the Volgograd district, the reference thickness of the A horizon of noneroded grey–brown soil is 15–20 cm and that of the B_1 horizon is 31–49 cm on the slopes of the Tinguta dry valley. Here the A horizon is completely washed away on severely eroded soils and the B_1 horizon is 8–19 cm deep (Lidov and Orlova, 1970). Detailed mapping of soil horizon depth transformation makes it possible to estimate the volumes and rates of erosion for the experimental sites and small catchments with Chernozem soils during the period of intensive agriculture (Table 1.8.3).

TABLE 1.8.3 Soil loss for the period of intensive agriculture, estimated with the method of soil horizon transformation (after Azhigirov *et al.*, 1992)

Basin	Area, (ha)	% of arable land	Soil loss volume (m^3)	Erosion rate (mm yr^{-1})	District
Veduga Creek	7034	70	4026	0.67	Voronezhskaya
Malyi Kolyshley River	11775	75	19017	1.26	Saratovskaya
Gor'kaya dry valley	9235	30	1863	0.23	Stavropol'skiy
Large Pogromka River	22420	72	10477	0.52	Orenburgskaya

TABLE 1.8.4 Effect of long-term agriculture on the humus content of Chernozem soils

		Humus content (%)				
			Ploughed fields			
Soil horizon	Depth of horizon (cm)	Virgin lands	12 years	37 years	52 years	100 years
A	0–12	9.4	7.8	7.3	5.9	5.5
A	12–25	6.6	7.5	7.2	5.7	5.3
A	25–35	5.9	6.2	5.8	5.2	5.2
B_1	50–60	3.8	4.5	4.5	4.1	4.2
B_2	140–150	1.3	1.2	1.4	1.7	1.4

Intensive agriculture has resulted in the loss of fertility of soils, increased erosion, changes to the microflora and chemical composition of the soils because of changed vegetation and altered soil water conditions. One of the most important changes has been dehumification, reducing both the soils, agricultural productivity and its resistance to erosion. Grinchenko *et al.* (in Kaurchev, 1989) showed that, during ploughing, the humus content is reduced in Chernozems and is distributed more evenly with depth in the humus and illuvial horizons (Table 1.8.4). Priputina (1989) compared the humus contents of Chernozems of the Russian Plain determined by Dokuchaev at the end of the 19th century with those of the present [maps showing these contents for the two periods have been published by Alayev *et al.* (1990)]. Priputina (1989) showed that the eastern part of the Russian Plain experienced high losses of humus of 4–10% after 100 years of agriculture. Losses of 1–4% occurred in the western part. This pattern is explained by the more intensive erosion processes in the eastern area, leading to further erosion as the erodibility of the dehumified soils increased.

1.8.3 GULLY EROSION

1.8.3.1 Gullies: Distribution in the Territory

The territory of European Russia and Byelorus was divided (Litvin *et al.*, 2003) into the following five belts according to the genesis and the density of gullies (Figure 1.8.2a):

1. The belt of contemporary natural gully thermo-erosion (erosion of the frozen ice-containing soil by both thermal and mechanical action of water). The density of such thermo-gullies (the gullies where thermal destruction of ice inter-layers in soil is of the same importance as mechanical erosion) can locally reach >100 gullies per 100 km^2. Near towns, quarries and gas and oil fields, the natural instability of the landscape with the permafrost is increased by human impact, and the rates of initial gully growth can become catastrophic, up to several hundred metres per year.
2. The belt where gullies represent extremely uncommon and isolated phenomena (<2 gullies per 100 km^2) on nontilled or little tilled land with flat or rolling relief in the northern (>57–58° N) part of the forest zone or low-lying land with valleys <10 m deep (such as Poles'ye).
3. The belt of low gully density varying between 2 and 25 gullies per 100 km^2 over most of the area. Such areas have low relief with forested flat interfluves. They occupy the forest zone south of 57–58° N, part of the Dnieper lowland plain, the wooded upland flat areas of the Smolensk and Middle Russian Uplands and part of the Oka-Don plain. In the southern part of the forest zone the density of gullies can reach 25–50 per 100 km^2. Gullies in the forest were formed during the periods of much broader extension of tillage of the former arable lands.

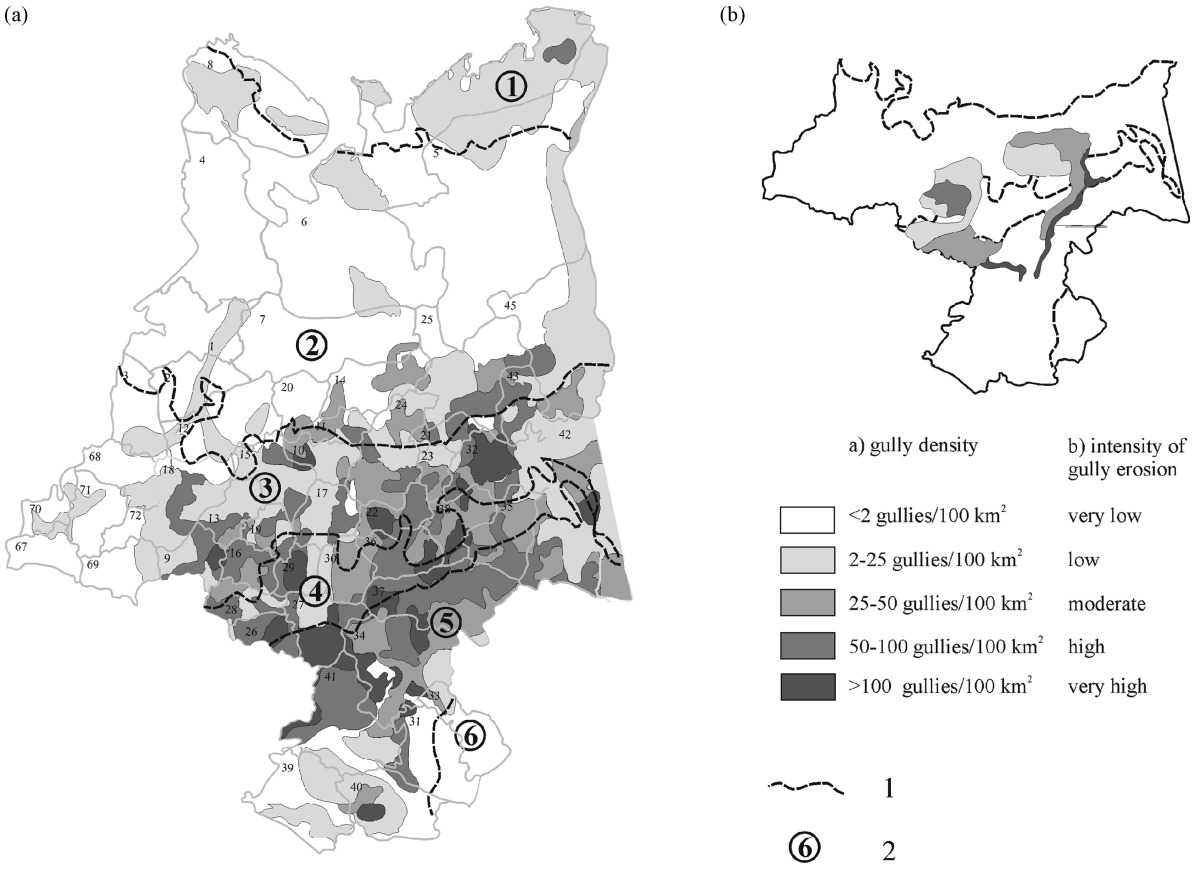

Figure 1.8.2 (a) Distribution of gullies on the Russian Plain, showing contemporary density (number of gullies per 100 km^2) and (b) categories of gully erosion intensity in the1930–40s (after Kozmenko, 1954). Key: 1, boundaries between vegetation zones; 2, vegetation zone index: (1) tundra (2) taiga (3) mixed and broad-leaved forest (4) forest–steppe (5) steppe (6) semi-desert

4. The principal belt of gullying in the forest–steppe and steppe zones. The main human factor in gully formation here is tillage of almost the entire area. Gullying is also fostered by natural conditions: substantial volumes of melt water and rainfall, relatively erodible loess subsoils and greater relative relief. When these areas were first cultivated, intensive tillage led to the formation of gully systems of the greatest extent and density, compared with other regions. Relative relief and land use differentiate the gully density within the belt. Areas with moderate gully density, 25–50 per 100 km^2, are typically in relatively flat ranges and uplands with shallow relief dissection (the Smolensk Hills, the north-western part of the Middle Russian Uplands), and also in rolling plains (the Tambov district, the Oka-Don plain, the western part of the Obshchiy Syrt). Areas of advanced development with relatively favourable natural conditions for gully formation are characterized by deeply dissected relief and high gully density: 50–100 per 100 km^2. Such regions include the central parts of the upland country: the Central Russian region and the Volga upland. Areas with very high gully density (>100 per 100 km^2) are found in a relatively small region in the middle of the upland country and along riverbanks, comprising <10% of the entire gullied land.

TABLE 1.8.5 Distribution (%) of gullies with different growth rates (after Kosov, 1970)

Land-use type	Total No. of gullies	Maximum annual (seasonal) growth (m)				
		<5	6–15	20–40	50–80	>100
Agriculture	269	50	25	15	8	2
Logging	15	25	18	25	7	25
Road building	17	15	25	30	25	5
Industrial development	19	20	20	25	10	25

5. The southern belt with very low gully density. This region includes the greater part of the Azov and Black Sea coastlands and the Caspian lowland.

The mean gully density in European Russia (3.8×10^6 km^2) is 28 gullies per 100 km^2. The gully net (Moryakova *et al.*, 1987) is formed by 1 045 600 gullies with a total length of 114 540 km, an area of 1040 km^2 and a volume of 3.5×10^9 m^3. In Byelorus (0.2×10^6 km^2) the gully density is 7 per 100 km^2, the net is formed by 14 500 gullies with a total length of 1700 km, with an area 16 km^2 and a volume of 0.054 km^3. These gullies have a length of >70 m and were formed mainly during the period of intensive agriculture (the last 300–400 years).

Kosov (1970) collected more than 300 measurements of gully growth rates in the European part of the former USSR for various land-use types (Table 1.8.5). About 45% of these data show gully growth during 1–5 years, 35% up to 10 years and the others for longer periods up to 170 years. The gullies on arable land are characterized mainly by medium rate of growth (50% of the gullies have a maximum growth rate of <5 m yr^{-1}). Catastrophic rates (>100 m yr^{-1}) of gully development are more typical for the areas of forest logging and industrial development.

1.8.3.2 Changes in the Rate of Gully Erosion

In the development of gully erosion, the same stages can be seen as in slope erosion. Using data from the chronicles of the 12–14th centuries and land registries for the 15–17th centuries, Sobolev (1948) noted severe linear erosion in towns and villages of the forest zone. Moryakova (1988) dated > 500 gullies in the Sod-Podzol soil region with the help of organic carbon content in the initial soils in the gullies. These data show five main periods of intensive gully growth with the maximum rate of gully formation in 1860–1910, when ~24% of now existing gullies were formed (Table 1.8.6).

The period of the fastest development of gullies within the forest–steppe zone of European Russia was the second half of the 19th century. Massal'sky (1897) used responses to his special questionnaire from correspondents throughout European Russia to obtain the first overview of the extent of gully erosion in the Chernozem Belt of European Russia. The highest intensity of gullying coincides with the areas of

TABLE 1.8.6 The main stages of gully formation in the Sod-Podzol soil belt (after Moryakova, 1988, with additions)

Period	% of gullies formed during the period	Volume of the gullies in 1970 (10^6 m^3)	Rate of gully formation (% yr^{-1})
1970–1910	9.0	16.5	0.15
1910–1860	24.2	44.4	0.48
1860–1730	40.4	74.2	0.31
1730–1600	21.2	38.9	0.16
1600–1500	5.2	9.5	0.05

historically early cultivation within the Chernozem zone (the Tula and Kursk districts). Two other periods with the growth of new gullies were registered in the forest–steppe and steppe zones during the late 19th and the mid 20th centuries. They were connected with cultivation of virgin lands, beginning from the end of the 19th century and up to the 1950s, and in some areas also with the restarting of cultivation after World War II. An attempt to compile a map of the gully regions (Figure 1.8.2b) was undertaken by Kozmenko (1954) for areas of the Middle Russian uplands and the Volga valley with the most sharply dissected relief. The data on gullying relate to the 1930–40s.

The tendency towards decreasing gully erosion rates during the second half of the 20th century is noted for all European Russia. According to field observations (Butakov *et al*., 2000), it decreased by 2–3-fold compared with the data for the beginning and middle parts of the century, collected by Kosov (1970) (Table 1.8.5). The most recent observations by Rysin (1998) in the Udmurtiya show mean gully annual growth within the range 2.1–2.2 m during the last 40 years. The maximum measured rate for a 15-year period was 40 m yr^{-1}.

1.8.4 SEDIMENTATION IN SMALL RIVERS

1.8.4.1 Spatial Distribution of Sedimentation Types

Field studies and map analysis make it possible to pinpoint typical forms of sedimentation in small rivers (Litvin *et al*., 2003). Their spatial distribution allows the classification of European Russia and Byelorus on the basis of combinations of natural and human-induced conditions. The following areas can be distinguished (Figure 1.8.3a):

1. Areas with predominant meandering rivers preserved in their natural, nonsedimented state with firm, well-defined banks and a dry flood plain. This area is thinly populated and little cultivated, being in the forest zone. Mean channel gradients of 0.2–0.8 ‰ ensure the transport of suspended sediments to the river mouth.
2. Areas in which rivers with swampy floodplains predominate: the rivers flow in wide relict valleys with very low gradients (0.05–0.15 ‰). The configuration of channels in swamps is highly erratic. Their width and depth change within very broad ranges (15–20-fold), and sometimes a channel disappears and water seeps across the swamp. Natural swampland is very vulnerable to human-induced sedimentation.
3. Areas with both sedimented and nonsedimented rivers. Here incipient sedimentation in the channels of creeks adjoining major cropland and farming areas occurs, while creeks and rivers of the same size flowing through forests and flood plains remain in their natural state.
4. Areas in which creeks are mostly sedimented, while small rivers remain in their natural state. These conditions occur in the south of the forest zone and in the forest–steppe zone, where arable land occupies $<70\%$ of total catchment area. Most sediment from the slopes reach creeks up to 20 km long, where large-scale sedimentation occurs. This reduces deposition in the watercourses of the small rivers. Hence the creeks and flood plains serve as a buffer between the slopes and the rivers.
5. Areas with sedimentation of all small rivers and some of the medium-sized rivers. In the steppe zone, under conditions of intensive tillage of catchments, heavy water use, regular droughts and sharp flow peaks, the sediment yield from slopes can reach small and medium rivers. The result is that an ordinary channel spreads into a swampy network, in which the old channel is overgrown with reeds and marked only by firm, dry banks.
6. Areas with sedimentation of swampy floodplain-type rivers.
7. Areas of local internal drainage with very low drainage density and also riverless areas.

These areas correspond broadly to the natural landscape zones. Areas with no sedimentation coincide with tundra and taiga with their high runoff coefficients; those with both sedimented and nonsedimented rivers tend

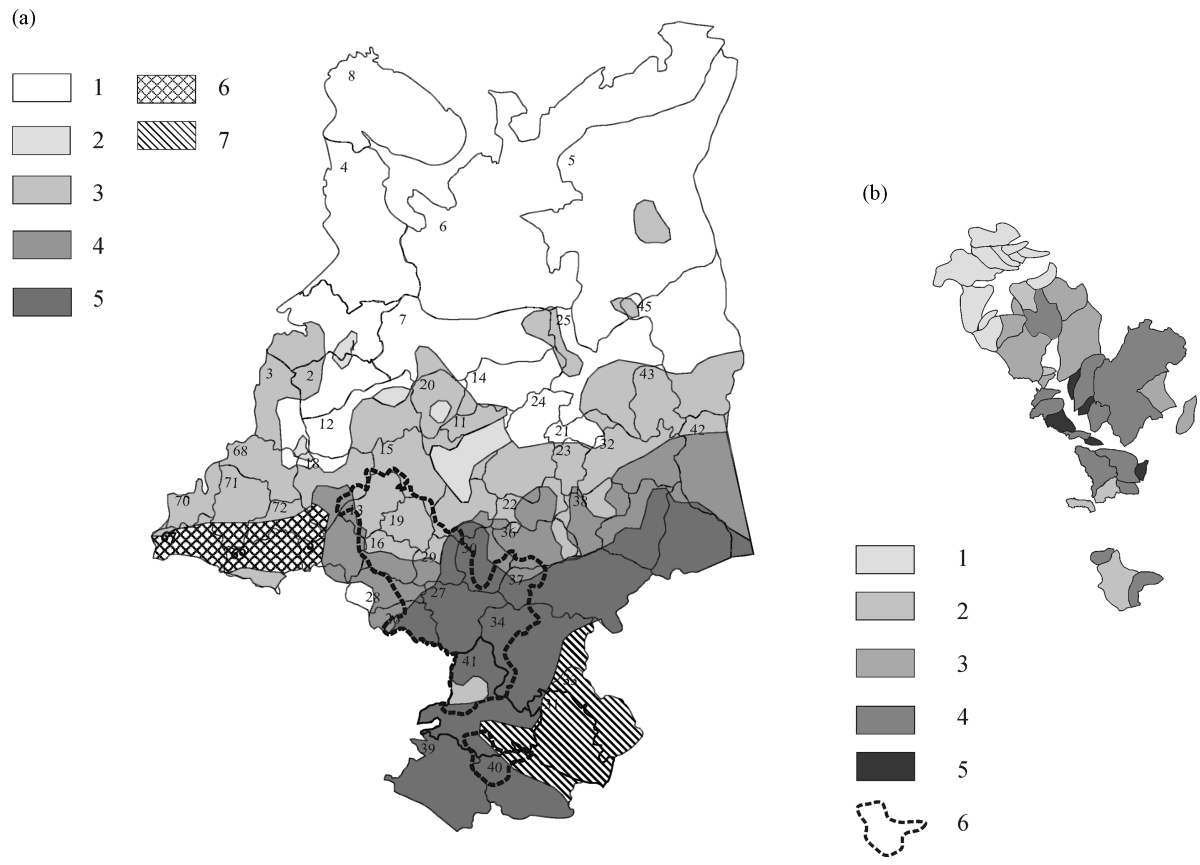

Figure 1.8.3 (a) Distribution of typical forms of sedimentation in small rivers in European Russia and Byelorus and (b) length of river net in a number of river basins in the middle of the 20th century as a percentage with respect to that at the beginning of the 19th century. *Key*: (a) see text; (b) 1, 90–110%; 2, 75–90%; 3, 60–75%; 4, 45–60%; 5, 25–45%; 6, outline part (b) on part (a)

to be related to the mixed and deciduous forest zone; those with sedimentation in the upper reaches of the rivers often correspond to the forest–steppe; heavily sedimented rivers are found in the steppe zone with low runoff coefficients; and inland drainage areas coincide with the arid steppe and semi-desert zones. At the same time, however, the outlines of these areas are more complicated than those of the landscape zones, and their limits frequently do not coincide with those of the latter. This may be because the type and level of the economic activity do not correspond to the geographical or terrain zones (as, for example, the penetration of agriculture into the taiga), and because of the azonal geological and geomorphological factors. The latter determine the shapes of the longitudinal profiles of rivers, the values of local slope gradients and the erosion and sedimentation capacity of watercourses. Areas, shaped mainly by neotectonics and geomorphology with swampy floodplain-type rivers, are scattered sporadically over all regions.

1.8.4.2 Stages of Aggradation in the River System

Permanent watercourses are sensitive to changes in climate and land use. The hydrological and sedimentological regimes of small rivers in European Russia and Byelorus are controlled by changes in the forest cover

and proportion of arable land in their catchments. Contemporary data (Golosov and Panin, 1998) show that tillage of up to 30% of the catchment area affects only the water runoff and sediment yield, without reducing the length of the river system owing to sedimentation. Data from 130 sites on 75 rivers with basin areas $<100\,km^2$ located in the middle part of the Volga river basin demonstrate that the deposition rate on the floodplain depends on the area of arable land in the catchment. The total thickness of sedimentation during the agricultural period is $\sim$1 m for basins $<$20% forested and close to zero in the completely forested basins (Kurbanova and Petrenko, 1990). Acceleration of floodplain aggradation is marked even for large rivers. Archaeological data show that aggradation rates for the period 2500–200 years ago were $\sim$0.6 mm yr^{-1} in the Middle Oka River floodplain and they increased to 6–6.5 mm yr^{-1} in the last 200 years (Glasko and Folomeev, 1981). Massal'sky (1897) noted that the Svirnya River (Don River tributary) was prone to sedimentation and some late 18th century coins were found in sediments at 1 m depth. The thickness of sedimentation was estimated at the bottom of 11 small valleys with basin areas of 5–40 km^2 in different regions of European Russia (the Middle Oka, the Upper and Lower Don, the Lower Volga, the Ural River and Stavropol' Region). It ranges from 1.0 to 2.8 m, with mean aggradation rates of 3–38 mm yr^{-1} for the period of intensive agriculture (50–350 years) (Golosov *et al*., 1991).

The spatial distribution of aggradation in small rivers was estimated on the basis of measurements of the length of the permanent stream net. A comparison of the 1:420 000 scale map of 1826–42 and the 1:300 000 map of late 1940s–early 1950s was made (Golosov and Panin, 1998) from the Upper Oka River basin in the north to the Kalaus River basin in the south (Figure 1.8.3b). During this 100-year interval there was no essential change in the length of permanent streams in humid landscapes of the southern part of the forest zone. Some rivers with densely forested catchments slightly increased the extent of the river system, owing to the process of incision into water tables in formerly dry valleys. The process of river shortening becomes evident towards the south-east of the forest zone and reaches high values (decrease of the river net length by $>$50%) in the semi-arid regions (the southern forest–steppe and the northern steppe). Relief, ground water, soils and rock type affected the spatial distribution of river net reduction. The length of the river net decreased by 42% in the Middle Russian Upland and by 31% in the Oka-Don Lowland (both are located in the forest–steppe zone). In the Medveditsa River basin, where the right-hand tributaries are fed by significant volumes of groundwater, the reduction of the left-bank tributaries was 21% and that of the right-bank tributaries only 9%. The high rates of sedimentation in the rivers of the Khoper basin can be explained by the high soil erodibility (sand and silt).

The volume of sedimentation in rivers of different sizes may be estimated using data on catchment erosion and sediment delivery ratio. These estimates show (Sidorchuk, 1995) that in the last 300 years most sedimentation has been concentrated in the floodplains and channels of dry valleys and creeks 10–25 km long. The volume of sediment diminishes from west to east, and also to the north and south of the central zone of maximum sedimentation. This zone embraces the Oka basin (deposition thickness $h = 2.7$–3.1 m) and the Vyatka and upper Kama basins ($h = 1.9$–2.7 m). North-west of this zone the depth of sediment declines to 1.1–2.4 m (Upper Don and Volga basins) and to the south-west to 0.5–2.3 m (Don and Middle and Lower Volga basins). The measurements in the deltas of the major rivers show that only 6–7% of eroded soil is transported to the seas, the main part being sequestered in the fluvial system (Sidorchuk, 1995).

1.8.5 OTHER SOIL LOSS PROCESSES

1.8.5.1 Wind Erosion

Wind erosion prevails on arable land in the south-eastern part of European Russia (see Figure 1.8.1), where silt soils on the hilltops and leeward slopes are easily dried and deflated by winds (Larionov, 1993). The frequency and intensity of wind erosion events increased with the expansion of agriculture in this region: nine 'black'

storms were observed in the 19th century, five during the first 30 years of the 20th century and 25 from 1940 to th 1970s. Between 3 and 30 less intensive 'dust' storms with the wind velocities up to 10–30 m s^{-1} occur each year. The most catastrophic were the 'black' storms of 1969–70, which happened when some fields, not protected by the forest buffer strips, lost 0.26 cm of soil on average (and up to 70 cm in some places). Many forest buffer strips were completely buried with soil and formed elongated hills 2–3 m high and 30–50 m wide and these remain in the landscape. The soil dust from these storms was observed in the Ukraine and Moldavia, Sweden and The Netherlands (Larionov *et al*., 1996). Annual soil loss due to wind erosion is estimated as 5–40 t ha^{-1} in the Northern Caucasus and 5–22 t ha^{-1} in the Lower Volga region (Larionov, 1993).

Wind erosion on pasture is associated with light sandy soils and overgrazing. It is common in the tundra zone, where reindeer overgrazing leads to the formation of active sand dunes around towns and villages. The same type of movable sand destroys pastures in the Kalmykiya and in the Lower Don region as a result of sheep overgrazing.

1.8.5.2 Tillage Erosion

The influence of tillage (mechanical) erosion on the fields of European Russia and Byelorus is evident. Most of the convex interfluvial areas on the fields show truncated soil profiles, often with B or C horizon exposed on the field surface. Narrow bands of accumulated soil 10–20 cm high mark the field edges. This process is more obvious on Sod-Podzol soils (Zaslavskiy, 1983). Tillage erosion is combined with intensive water erosion on convex–concave slopes. On such slopes, stable systems of ephemeral gullies are formed during the melt period or summer rainfall. When ploughs and harrows level the field, the trenches of ephemeral gullies are filled by loose topsoil from surrounding areas, and therefore soil profiles become thinner. Melt water flow or intensive rainfall renews the incision of the ephemeral gullies and removes most of their infill from the field. The cycles of levelling by tillage and dissecting by erosion lead to general intensive soil loss. Observations on the soil profile truncation on one such field (170 ha) in the Stavropol' district showed a decrease of reference chernozem soil depth ($A + B_1 = 80$–90 cm) during the last 70 years to 36–57 cm on the inter-gully areas and 10–15 cm in ephemeral gullies. The mean annual soil loss from combined tillage and water erosion amounted to 58 t ha^{-1} at this site (Belyaev *et al*., 2005).

1.8.5.3 Soil Loss with the Harvest

One of the specific types of soil loss is mechanical removal of soil from fields with the harvest, mainly with potato and root crops (sugar beet, carrot and radish). Zaslavskiy (1983) estimated this loss as 5–10% of the harvest weight. Belotserkovskiy and Larionov (1988) showed by direct measurement of adhered soil from potato and beet in the Kaluzhskaya district that the soil loss with harvest in 1975–80 was 2.5 t ha^{-1} with potato and 2.3 t ha^{-1} with beet. The measured soil delivery by melt water flow from different fields of the same farm was 0.08–2.0 t ha^{-1} per spring season 1982–89. The reports of one of the crop warehouses in Moscow, where root crops were washed before being delivered to the market, showed a lower proportion of soil in the harvest than the above measurements made near the field (Table 1.8.7). This difference is related to the distance from the field to Moscow and partial loss of adhered soil during transportation. Nevertheless, even this underestimation of soil loss with potato harvest ($\sim$0.6 t ha^{-1}) gives $\sim 1.5 \times 10^6$ t of annual soil loss from 3×10^6 ha of potato fields in European Russia and Byelorus.

1.8.5.4 River Bank Erosion

River bank erosion is mainly a natural process in European Russia and Byelorus. The total length of the rivers is 711 855 km and 93% are sinuous or meandering, with 30–40% the banks affected by erosion. The rate of

TABLE 1.8.7 Soil delivered to Moscow with potatoes in 1985 (after Belotserkovskiy and Larionov, 1988, simplified)

District	Potato + soil (t)	Soil (t)	Mean harvest in 1981–85 (t ha^{-1})	Mean soil loss with the harvest (t ha^{-1})
Russia				
Bryanskaya	247.2	3.9	11.6	0.2
Orlovskaya	62.1	1.2	9.1	0.2
Ryazanskaya	1096.7	17.7	9.0	0.1
Moskovskaya	5085.1	346.6	12.7	0.9
Tverskaya	430.9	7.4	10.3	0.2
Kaluzhskaya	4451.5	472.9	10.3	1.1
Byelorus				
Brestskaya	2794.3	50.3	17.1	0.3
Grodnenskaya	847.9	5.9	15.6	0.1
Minskaya	6168.2	164.9	15.3	0.4
Vitebskaya	655.9	11.8	13.6	0.2
Gomel'skaya	814.6	14.8	15.6	0.3
Mogilevskaya	820.9	18.6	14.9	0.3

river bank erosion is controlled by discharge and slope. For small and medium rivers of the Volga and Don basins it increases with the river size (Table 1.8.8). On the large rivers, with mean maximum discharge (MMD) $>4000\,m^3\,s^{-1}$, the annual rate of bank erosion can exceed 6–10 m (Chalov, 1994): for the Lower Vychegda River it is 12–40 m yr^{-1}, for the Lower Don it is >6 m yr^{-1} and for the Lower Volga it is >10 m yr^{-1}. Eroded particles are mostly deposited within a river channel on the bars and lower floodplain, so that the river channel width remains stable in the long run. For example, on the Lower Terek River the mean rate of bank erosion in 1932–72 was 2.7 m yr^{-1}, with local extremes of 10–15 m yr^{-1}. Such a rate corresponds to sediment production of 0.8×10^6 t yr^{-1}. Sedimentation within the active belt of the river was also $\sim 0.8 \times 10^6$ t yr^{-1}, so that the budget of channel-forming particles was close to zero (Alekseevskiy and Sidorchuk, 1990).

1.8.5.5 Reservoir Bank Erosion

Bank erosion in artificial reservoirs is a purely human-induced process. Here steep profiles of the shore zone, wave height and regime after the reservoir filled with water are completely different from those on natural coasts close to equilibrium. The rate of abrasion is catastrophic and locally exceeds 200 m yr^{-1} in the initial period of reservoir formation, decreasing through time with the increase in the abrasion bench width. The reservoirs in Byelorus situated mainly in the forest zone are rather small: there are 130 reservoirs with a total volume of 2.45 km^3 and an area of 715 km^2. The length of the reservoir banks is 1300 km and ~25% of these are abraded by wave action. A stabilizing bench 12–30 m wide and 1.5–2.0 m deep appears after 15–20 years

TABLE 1.8.8 Distribution (in % of the river length) of the rate of river bank erosion (after Kamalova, 1988)

MMD ($m^3\,s^{-1}$)	River bank erosion rate (m yr^{-1})				
	<0.5	0.5–1.0	1.0–2.0	2.0–3.0	3.0–4.0
<300	13	28	53	5	1
300–1000	3	17	39	38	3
>1000	9	19	16	12	44

in reservoirs with a stable level regime and after 25–30 years in reservoirs with a variable regime. The loss of land around such reservoirs is ~5000 ha (Shirokov, 1991). A similar regime characterizes small reservoirs in European Russia.

Soil loss processes at a different scale are observed in the giant reservoirs on the largest lowland rivers of European Russia, the Volga, Kama and Don. The total volume of 16 reservoirs of the Volga–Kama system is 197.3 km^3, with a combined area 2.8×10^6 ha. Arable land constituted 11% of this now-flooded area, 38.8% was pasture and 36.8% was forest (Vendrov, 1979). From 10 to 40% of the shoreline of these reservoirs is intensely attacked by waves. The rates of bank erosion were 10–50 and up to 120 m yr^{-1} during the first 16 years of the life of Rybinskoye reservoir, 25–40 and up to 50 m yr^{-1} for the first 13 years at the Gor'kovskoye reservoir and 70–90 and up to 210 m yr^{-1} for the first 10 years of the Volgogradskoye reservoir (Finarov, 1986).

1.8.6 CONCLUDING REMARKS: LAND-USE TRENDS OF RECENT DECADES

The most recent information about erosion processes in European Russia and Byelorus belongs mainly to middle 1980s. After that, radical changes in the political situation and economy began in the USSR. The data collected by scientific institutions and government authorities during the last 10–15 years are fragmentary and uncertain. Federal and regional land-use and soil-conservation policy is unclear and changeable. Federal statistics of the Russian Federation (Anon., 2005) show dramatic land-use changes. In 25 800 large collective farms and state agricultural complexes, which used 86–93% of the land, the area of arable land decreased by 20%, the area of sowing decreased by 42% during 1990–2004. The volume of agricultural production decreased by 60% during 1990–2000. Changes in the type of management (87% of the former collective farms and state farms became stock companies and cooperatives in 1994) and decreases in food imports in 1998 caused some (11–12%) increase in the volume of production during 2000–2004. A considerable amount (30–60%) of food (mainly vegetables) was produced by both the urban and country populations (34–35 million families) on private lots (with area ~0.09–0.4 ha each), which overall occupy 7.4% of arable land. Individual farmers, who used about 11–12% of the arable land in 2004, produced only 6% of total agricultural production. The number of such farms, with a mean area of 73 ha, increased sharply in the first years of economic changes (from 100 in 1990 to 183 000 in 1993 and 280 000 in 1996), then slightly decreased and has now stabilised at the level of 260 000–265 000 farms.

These statistics show that the main land user (at least 86% of the land) is still large farms (4000 ha on average) with a collective type of land use. The pattern of the fields (their length and inclination) did not change significantly. About 25% of the fields are not used and are covered at present by weeds and scrub. Water erosion is negligible there. The market dictates the crop rotation on the other parts of the land, and the land conservation methods of management are out of use. Often a mono-crop culture (such as sunflower) can be cropped for several years of high prices for this type of production. Water erosion rates on such fields could be significantly higher than in previous years.

The erosion pattern on land used by individual farmers is unclear. Most of these fields, cut out of the large collective farms, are situated on the poorest and most eroded soils and on the slopes. Many of these farms are now abandoned and not used for agriculture. Some of them are exploited without any care about erosion processes and represent potential spots of significant soil loss.

The plots of citizens' private land are mainly used as vegetable gardens with an organic type of farming. Erosion on arable land of this kind is absent and soil fertility increases rapidly.

We can conclude that the current situation with erosion processes in European Russia and Byelorus is uncertain. The system of land ownership and management is changing slowly. One of the main effects of this process is a substantial decrease in the land under the plough and, therefore, a decrease in the extent of erosion

processes (by 25% at least). Simultaneously, new spots of locally high erosion rates could appear owing to the increase in the proportion of farms with specialized unvarying crop rotation. People with no experience of land husbandry are taking up farming. Previous state departments for soil conservation do not work and the new ones are not yet properly organized. If a repetition is to be prevented of the general degradation of soils that occurred in the late 19th century, following the abolition of serfdom, it is essential that a well-considered state policy of conservation education be pursued, and that a body of laws be designed to promote farming techniques which conserve soils and water resources. A first step in this direction is the content of Chapter 2, 'Land Conservation', of The Land Code of the Russian Federation of 2001. This chapter declares the duty of landowners and land users to keep soil fertility and to prevent water and wind erosion. This declaration shows the necessity for a special branch of land-use legislation, as part of general environmental legislation (Bogolyubov and Minina, 2002).

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1.9

Poland

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1.9.1 INTRODUCTION

Poland (312 685 km^2) is predominantly a lowland country (>75%). The altitude ranges from 1.8 to 2499 m, with an average of 173 m. About 90% of the country is between 0 and 300 m above sea level. Generally, topographic features are arranged in belts parallel to latitudes. Higher areas are located in the south (Carpathian and Sudety mountains and the belt of Polish uplands) and built from marine deposits of various ages (from Paleozoic to Tertiary). Flysch (sandstone and shale) prevails in mountain areas and calcareous and sandy limestones in uplands. Parts of the mountain foreland and uplands (up to 30%) are covered by loess. Central and northern parts of Poland are lowlands with glacigenic deposits (boulder clay, sands and gravels). In the north, young glacial landscapes dominate with numerous small hills (lakelands), whereas the plains of the central part are built from deposits of older glaciations. The largest area of Poland is occupied by soils characteristic of mixed forests (Luvisols and Cambisols). Fairly large areas are also occupied by Podzolic soils, developed under coniferous forests on sandy deposits. River valleys contain alluvial soils of different textures. Locally, Rendzinas and Chernozems are present in southern part of country and Regosols in mountains.

The climate is moderate and affected by both maritime and continental air masses. The average annual temperature is about 7.5 °C with a range of average monthly values from about –3 to 18 °C. Annual precipitation in the lowlands is 500–550 mm, in the belt of Polish Uplands and Lakelands 600–700 mm and in the mountain area 700–1000 mm. Most of the precipitation is in the form of rain (about 80% with its maximum in July). Snow cover lasts from 40 days in the west to 100 days in the north-east. Its maximum thickness reaches 20–30 cm in the west and 70 cm in the north-east. In the majority of mountain areas, snow cover lasts over 100 days and its maximum thickness exceeds 100 cm. The annual erosivity index (R) calculated for the eastern part of Poland ranges from 426 to 968 MJ mm ha^{-1} h^{-1} with lowest values in the north and the highest

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in the south (Banasik and Górski, 1993). For south-west Poland, Licznar and Rojek (2002) obtained an annual value of 637 MJ mm ha^{-1} h^{-1}. The majority (64–74%) of the value occurs in the period from June to August. Both monthly and annual values are characterized by high variability. Heavy rainfall has a higher probability in the mountain regions and the belt of Polish Uplands and to lesser extent in the Lakelands and along the Baltic coast (Kostrzewski *et al*., 1992; Starkel, 1995, 1998; Rodzik and Janicki, 2003). The largest recorded discharge was 37 m^3 s^{-1} km^{-2} on 18 May 1996 as a result of a rainfall event of 120 mm in 2 h (catchment of 1 km^2 near Cracow) (Niedbała and Soja, 1998). Usually, the duration of erosive rainfalls does not exceed 30 min (but sometimes lasts up to 2 h) with intensities of 0.5–3.0 mm min^{-1}. The recurrence period of a 30-mm event is estimated at 2 years, a 60-mm event at 10 years, a 115-mm event at 100 years and a 220-mm event at about 500 years (Wierzbicki and Bartkowski, 1969).

Most of the country is occupied by arable land (45%). Orchards occupy 1%, grasslands 13%, forest 29% and others 12%. The highest percentage of arable land is concentrated in the Polish plain and loess areas of the Polish Uplands, with the largest percentage of forests in the in northern and southern parts of the country.

1.9.2 HISTORICAL EVIDENCE OF EROSION

The first traces of agricultural activity in Poland were in the Neolithic period. In contrast to earlier opinion that human pressure was too insignificant to stimulate erosion in this period, sediment analysis by Starkel (1988) near Cracow and by Śnieżko (1995) in dry loess valleys of Lower Silesia confirms that some of the human cultures affected changes in the sedimentation regime. More erosion forms date from the Bronze Age (2300–1300 BC) due to an expanding population and farming activities. In the foreland of mid-mountain areas, fluvial processes affected by climatic factors started to be influenced by humans when the cultivated areas of small catchments increased to 60% (Klimek, 2002); this led to an increase in alluviation in the stream at Cracow (Klimek, 1988). Initial phases of gully development were found by a Polish–German team in the south-east (Zgłobicki *et al*., 2003), alluvial fan formation in the south-west (Zygmunt, 2003) and anthropogenic colluvium in northern Poland (Sinkiewicz, 1998). Before the establishment of the Polish state in the 10th century, phases of establishment and abandonment of particular sites by migrating peoples are reflected in traces of erosion seen in alluvial fans. More frequent effects of human activity began in medieval times (Starkel, 1988). Extreme floods were recorded from the valleys of middle and western parts of the Sudety mountains in the period 1310–1400 (Klimek, 2002). In contrast, more frequent floods in the Vistula basin occurred in the 14 and 15th centuries, and especially from the second part of 16th until the 19th century (Maruszczak, 1997). In Maruszczak's opinion, most of the present gully systems started to develop from the 14th century, with maximum rates at the beginning of the 17th century (Zgłobicki *et al*., 2003). This coincides with the Little Ice Age. More frequent extreme events corresponding to climatic change were found in earlier times by Starkel (1986). Increased human pressure was reflected in the frequency of floods on the upper Vistula river. In the 19th century floods took place once every 4.2 years, and at the beginning of 20th century once every 2.8 years (Maruszczak, 1997). An interesting record of denudation processes is given by Borówka (1990). He established the following denudation rates per 100 years: 1.76 mm (late glacial period), 0.027 mm (Holocene–Christian era), 0.25 mm (10th century), 1.75 mm (10–14th centuries) 1.15 mm (15th century) and 4.5 mm (20th century) in a closed basin located in the Polish plain.

1.9.3 STUDIES AND ASSESSMENT OF EROSION

Studies of soil erosion have a long tradition in Poland. In 1928, Bac performed the first measurements of erosion based on comparisons of relative altitude changes. He found that the average soil loss on cultivated

loess slopes of 10%, over a period of 43 years, was about 5 mm annually (Bac, 1928). Intensive studies of soil erosion started after World War II. At the beginning of the 1950s, an erosion risk assessment map was developed and measurement of erosion in catchments began (Reniger, 1950). At the same time, soil conservation practices were developed (e.g. Ziemnicki, 1955), studies of soil translocation due to tillage (e.g. Czyżyk, 1955) and regular measurements of erosion processes (Gerlach, 1966) were initiated. The first runoff plots were established in Posorty (Mazury Lakeland) in 1956 (Skrodzki, 1972). Two years later, runoff plots were located in Szymbark (Carpathians), being part of an experimental station of the Institute of Geography and Spatial Organization of PAS (Gil, 1986, 1999). Although measurements on the former site were stopped, they are continuing on the latter. In 1984, Froehlich's group initiated studies with ^{137}Cs in the Homerka catchment (Froehlich *et al*., 1993).

A map of erosion risk assessment was initially developed by Reniger (1950) and systematically improved by Józefaciuk and Józefaciuk (1995). Based on the latter work, a map of erosion distribution is presented in Figure 1.9.1. According to Józefaciuk and Józefaciuk (1995), about 29.7% of the country is at risk of water erosion (with 9% at medium risk and 4% at strong risk). The most at-risk areas are in mountain regions, in the belt of the Polish Uplands and Lakelands. The assessment was based on topography, soil and rainfall analysis.

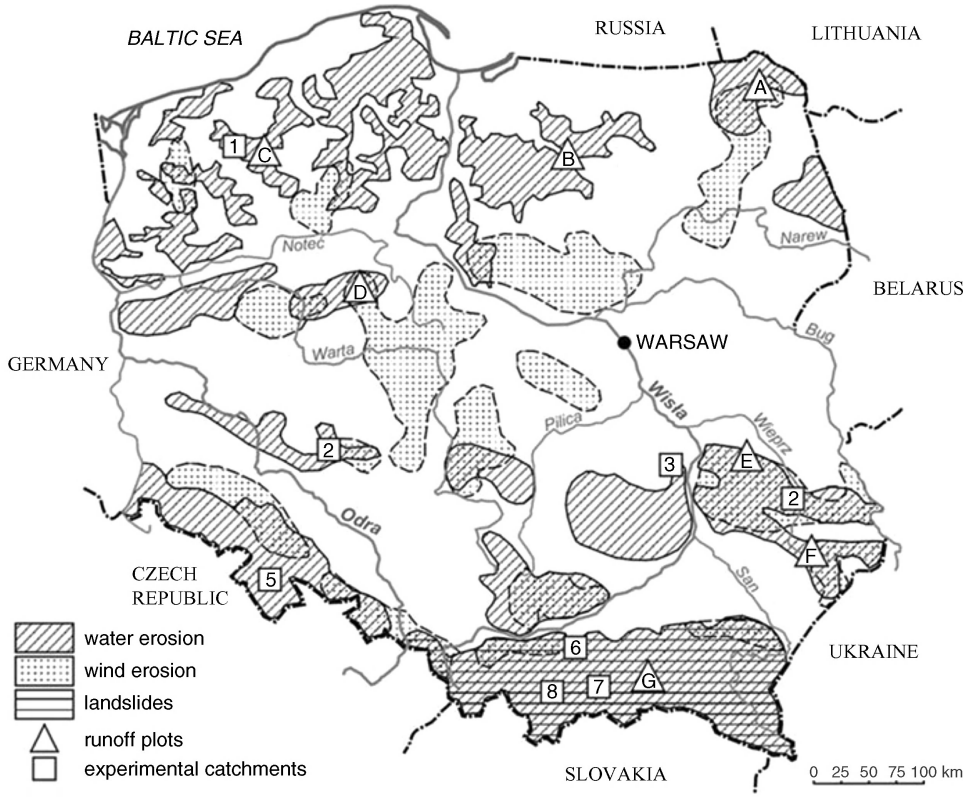

Figure 1.9.1 Areas of erosion risk in Poland (modified, after Józefaciuk and Józefaciuk, 1995). Experimental plots: A, Szeszupa; B, Posorty; C, Storkowo; D, Mokronosy; E, Czesławice/Bogucin; F, Guciów; G, Szymbark. Experimental catchments: 1, Storkowo; 2, Mielnica, 3, Zagożdżanka, 4, Niemienice and Wielkopole, 5, Wilkanów and Stara Łomnica; 6, Łazy; 7, Homerka; 8, Lubieńka and Kasinka streams

1.9.4 QUANTITATIVE EVALUATION OF EROSION AT DIFFERENT SCALES

Quantitative evaluation of erosion processes is extremely difficult. Fairly early it was recognized that soil loss measured at the catchment outlet and expressed per unit area of the catchment did not reflect the intensity of erosion inside the investigated area (Reniger, 1955). Later, this was also recognized for plot studies (Słupik, 1986). Generally, different studies showed high erosion rates when small contributing areas were considered, but with an increase in 'contributing' area erosion rates started to decrease. For example, in the Vistula basin, the sediment load was 97 t km^{-2} yr^{-1} in the upper part (mountain area), 9 t km^{-2} yr^{-1} in the upper-middle part, falling to 2 t km^{-2} yr^{-1} in the middle and lower parts. The total load extrapolated to the whole basin area is 7 t km^{-2} yr^{-1} (data compiled by Maruszczak, 1984). Without knowledge of the real contributing area, any comparisons among similar catchments or even plots of the same size should be treated with great caution.

1.9.4.1 Hillslope Scale

Usually, the highest erosion rates are found at the plot scale. In Poland, measurements on runoff plots were carried in a limited number of sites (Table 1.9.1). To study runoff events, plots of different sizes and located on various slopes were used. Generally, the period of measurement did not exceed 4 years, and long-term records were compiled only at Szymbark. Most of the soil loss occurred in summer, on plots with and without plants. Based on plot studies, erosion in winter (from November to April) is assessed at 0.9 t ha^{-1} (Gil, 1986, 1999) and 0.9–1.5 t ha^{-1} (Rejman *et al*., 1998). Plot studies in Szymbark have shown large variations in soil erosion between years, but without noticeable trends over a period of 20 years. Large difference in soil loss found on two loess sites could be related to short-term changes in rainfall pattern, as noted for a neighboring area (Rodzik and Janicki, 2002). Results of two plot studies were used to validate the USLE model. For loess soil, the experimentally derived soil loss was smaller by 2–8 times (Rejman *et al*., 1998; Rejman and Usowicz, 2002) and for loam by two times (Stasik and Szafrański, 2001). Over-prediction of soil loss with the USLE model seems to be connected mainly with short-distance transport of soil and deposition within the runoff plot (Froehlich, 1992; Rejman and Usowicz, 2002). In the former experiment, displacement was in the range 2–9 m (grassed slope of 14°) and in the latter, in the range 2–13 m (bare plots, 12% slope). The transport of soil for short distances could explain why, despite differences among plots, similar soil loss was found under cereals and potatoes in all sites (Table 1.9.1). Some of the experiments on slopes were carried with Gerlach troughs on 'plots' without side borders (Gerlach, 1966). In such cases, it is assumed that soil is transported from the slope divide and travels down the steepest gradient to the troughs. These studies were used to assess soil redistribution along slopes and showed that convex and usually upper slope segments are most eroded (Smolska, 2002; Stępniewski, 2002; Święchowicz, 2002).

Another method of erosion assessment on slopes which is still used in Poland is comparison of relative altitudes using reference points. For this purpose, transects are analyzed after periods of at least 20 years. This assessment takes into account not only erosion by water but also soil translocation due to tillage. The results do not differ too much from those of Bac (1928) and are in the range 4–5 mm yr^{-1}.

1.9.4.2 Ephemeral Forms

Specific erosion forms, characteristic of dry valleys, are episodic channels (summer ephemeral gullies) and rills (winter ephemeral gullies), being the effect of concentrated overland flow. The former are characteristic of heavy rainfall and the latter of abrupt snowmelt or prolonged rainfall. Usually, channels occur in cereals, their depth does not exceed 0.3 m and their width is up to 2–4 m. Teisseyre (1995) distinguishes two forms of channel, erosional (with low canopy cover) and depositional (where canopy cover induces sedimentation). After a runoff event in Lower Silesia, Teisseyre (1995) observed lowering of the ground surface within erosional channels by

TABLE 1.9.1 Characteristics and results of plot experiments

Location	Plot size (m)	Measurement period Start	Measurement period End	Annual precipitation (mm)	Slope (%)	Soil type	Land use	Soil loss ($t\,ha^{-1}\,yr^{-1}$)	Reference
Bogucin[a]	3 × 20	05/1998	08/2000	254	12	Silt loam (loess)	Bare plots	39.77	Rejman and Usowicz (2002)
Czesławice	3 × 20	07/1992	07/1995	592	9–10	Silt loam (loess)	Bare plots	10.00	Rejman *et al.* (1998)
							Cereals	1.24	Rejman (1997)
Mokronosy	6 × 40	11/1995	10/1998	554	4–12	Loam	Cereals	0.305	Stasik and Szafrański (2001)
Posorty[b]	7 × 120	01/1956	12/1967	637	25	Sandy loam	Along slope	15.13	Skrodzki (1972)
							Across slope	8.06	
							Grass	0.04	
Storkowo	4 × 42	03/1994	10/1996	687	9	Sandy loam	Bare plots	4.64	Szpikowski (1998)
							Cereals	1.90	
							Potatoes	19.21[c]	
Szymbark	10 × 60	11/1972	10/1981	863	18	Clay loam	Cereals	2.57	Gil (1986)
							Potatoes	21.84	
							Meadow	0.12	
Szymbark	10 × 60	11/1981	10/1990	803	18	Clay loam	Cereals	1.09	Gil (1999)
							Potatoes	34.27	
							Meadow	0.06	

[a]Period of analysis, May–October; rainfall data, only from runoff events.
[b]Cereals and fodder–beets analyzed together; results from 4 years in the period 1956–67.
[c]Soil loss on potato plots affected by rill erosion.

0.12 m and raising of the bed of depositional channels by 0.02–0.12 m on average. Similar values have been reported from another loessial areas in southern Poland. According to Teisseyre (1995), episodic channels can originate as a result of rainfalls of 10–40 mm with recurrence intervals of 2–3 years.

Figure 1.9.2 presents an example of an ephemeral channel system developed after heavy rainfall of about 60–70 mm in ~1 h.

1.9.4.3 Catchment Scale

Large numbers of studies have been carried out at the catchment scale (Table 1.9.2). Generally, catchment response depends on rainfall events. The most serious erosion caused by prolonged or extreme rainfalls was recorded in catchments in mountain areas (Froehlich, 1975; Rojek and Żmuda, 1992) and in western Pomerania (Kostrzewski *et al.*, 1994). Such events occur locally. From the cited studies, soil loss in Wilkanów ($519\,t\,km^{-2}$) was affected by one event of $150\,mm\,day^{-1}$, whereas such rainfall did not take place in the neighboring

Figure 1.9.2 Ephemeral channels and rills at Garbów catchment (Lublin Uplands), 16 September 1995. (Reproduced by permission of W. Zgłobicki)

catchment of Stara Łomnica (Rojek and Żmuda, 1992). For catchments located in mountain areas, Starkel (1986) distinguishes three types of rainfall with corresponding geomorphic responses. Intense rainfall (1–3 mm h^{-1} with amounts of about 100 mm) are related to linear erosion forms, surface runoff and wash, prolonged rainfall (2–3 days with amounts of 200–500 mm) to shallow mass movements and floods in river valleys and long-term rainfall (lasting several months) to landslides. The other factor affecting soil loss is connected with land use. The high percentage of arable land in the Wielkopole was reflected in high soil loss (up to 338 t km^{-2}), whereas the neighboring catchment of Niemienice with higher forest cover had a much smaller loss, despite similar precipitation (Pałys, 2001). A long record of erosion in loess catchments was given by Mazur and Pałys (1992). Over 35 years, extreme erosion events occurred six times (five times due to snowmelt) and the largest took place in 1956 (159 t km^{-2}). At the beginning of the observations, moderate erosion events from snowmelt (1.0–2.5 t km^2) were characteristic of the catchment. From 1969, the frequency of these events decreased. Generally, erosion took place in early spring and was caused by snowmelt representing 96% of total erosion. Also in other catchments in Table 1.9.2, this period, although to lesser extent, was characterized by the largest denudation coefficients. Such a distribution of annual denudation is disturbed by extreme rainfall taking place from late spring to early autumn and responsible for maximum soil loss.

Considering the catchment response, it should be mentioned that extensive and multidisciplinary studies have been carried out in catchments of Homerka (Froehlich, 1982), Parseta (Kostrzewski *et al*., 1994), Zagożdżonka (Hejduk and Banasik, 2002) and Łazy (Krzemień and Sobiecki, 1998). These studies concentrated on mechanisms of transport of dissolved and suspended material and analysis of water discharge. According to Froehlich (1992), cart roads are the main source of sediments, being responsible for about 80% of erosion in the catchment located in the mountain area. The share of sediment from cultivated fields is much lower owing to its location on terraces. Measurements of ^{137}Cs showed that the annual soil redistribution on the terraces is about 4 mm (Froehlich *et al*., 1993).

TABLE 1.9.2 Characteristics and results of chosen catchment studies

Site and location	Cachment size (km^2)	Measurement period		Annual precipitation (mm)	Soil	Land use		Denudation $t\,km^{-2}\,yr^{-1}$		Reference
		Start	End			Type	%	Mg	Avg.	
Wielkopole (Lublin Upland)	1.88	1987	1999	553	Silt (loess)	Arable land	80	338.0	59.9	Pałys (2001)
						Forest	14			
						Grassland	0			
Stara Łomnica (Sudety)	3.47	1988	1990	709	Clay	Arable land	32	13.3	6.4	Rojek and Żmuda (1992)
						Forest	41			
						Grassland	25			
Wilkanów (Sudety)	4.53	1987	1990	611	Clay	Arable land	20	561.5	162.7	Rojek and Żmuda (1992)
						Forest	34			
						Grassland	42			
Niemienice (Lublin Upland)	5.58	1987	1999	553	Silt (loess)	Arable land	10	1.8	0.4	Pałys (2001)
						Forest	86			
						Grassland	0			
Elizówka (Lublin Upland)	6.22	1956	1991	556	Silt (loess)	Arable land	85	159.2	8.0	Mazur and Pałys (1992)
						Forest	1			
						Grassland	2			
Mielnica (Lower Silesia)	6.67	1982	1994	638	Silt loam	Arable land	70	44.4	9.2	Żmuda (1998)
						Forest	6			
						Grassland	10			
Łazy (Carpathian Foothills)	22.4	1993	1996	659	Flysch	Arable land	39	59.7	—	Krzemień and Sobiecki (1998)
						Forest	41			
						Grassland	13			
Kasinka stream (Beskids)[a]	32.0	1967	1998	931	Flysch	Arable land	41	—	113.3	Lipski and Michalczewski (1998)
						Forest	44			
						Grassland	7			
Lubieńka stream (Beskids)[a]	48.7	1954	1998	931	Flysch	Arable land	48	—	167.7	Lipski and Michalczewski (1998)
						Forest	37			
						Grassland	13			
Parseta (Western Pomerania)	74.0	1986	1988	687	Sandy loam	Arable land	34	695	8.8	Kostrzewski *et al*. (1994)
						Forest	35			
						Grassland	17			
Kamienica Nawojowska (Beskids)[b]	239.0	1970	1971	947	Flysch	Arable land	36	1192	—	Froehlich (1975)
						Forest	43			
						Grassland	9			

[a]Calculated from siltation of reservoir.
[b]Denudation in the second year ($56\,t\,km^{-2}$).

1.9.4.4 Soil Cover Change

Erosion processes have affected the structure of soil profiles in agricultural areas. The truncation of profiles may reach more than 1.5 m on soils developed from loess (Janicki *et al*., 2002). Smaller reductions took place on Rendzinas, which are originally shallow, and on soils developed from young glacial deposits (Marcinek, 1994; Koćmit, 1992; Klimowicz and Uziak, 2001). Whereas Klimowicz and Uziak (2001) suggested that the

intensity of soil truncation was greatest a few decades after deforestation, Janicki *et al*., (2002) pointed out that the truncation rate increased rapidly at the beginning of the 19th century and has remained fairly constant since that time. Most work has concentrated on the distribution of eroded soils, showing their mosaic character and changes of soil properties (Turski *et al*., 1987; Licznar *et al*., 1992). The changes caused not only a reduction in yield by 20–40% with the ploughing-up of carbonate-rich loess (Rejman *et al*., 2001), but also significantly changed soil susceptibility to erosion (Rejman *et al*., 1998).

1.9.5 OTHER EROSION FORMS

1.9.5.1 Gully Erosion

Gully erosion occurs in about 18% of the country (Józefaciuk and Józefaciuk, 1995). A gully density above 0.5 km km^{-2} is regarded as a threshold of medium intensity of this erosion form. Areas of medium and higher intensities of gullies occur in 7.7% of the country and occupy 1.7×10^6 ha of arable land and 0.47×10^6 ha of forests. Gullies are concentrated in southern Poland with the most at-risk regions being the Western Carpathians (48% of the area with density above 0.5 km km^{-2}), the belt of east Polish Uplands (25–39%) and Sudety mountains (14–21%).

Despite natural conditions, the structure of farms is responsible for the development of such gullies. The southern regions of Poland are traditionally characterized by small fields and enormous numbers of cart tracks leading to these fields. Józefaciuk and Józefaciuk (1995) suggested that 52% of the total length of gullies is currently presently cart tracks. Analysis of gully development in the area of Kazimierz Dolny (Lublin Uplands), characterized by a gully density of 8–9 km km^{-2} suggests that the majority of gullies could be connected with human activity at present or in the past (Rodzik and Gardziel, 2004). For a system of medium-sized gullies, the total volume of material removed was assessed as 466 500 m^3 (Maruszczak *et al*., 1984). Only part of this amount was deposited in valley bottoms and outside the gully catchment. During an intense rainfall event of 102 mm, 5000 t were removed from this gully system in 1981 and 10 000 t from a neighboring one (Rodzik and Janicki, 2003).

1.9.5.2 Landslides

Active forms of landslides are seen on 100 000 ha (Ziętara, 1991). About 98% of that amount occurs in the Carpathians. The majority of landslides are located on forested slopes steeper than a 15°. Locally, landslides are found on the Baltic coast and on the banks of large rivers. In general, landslides developed at the end of glacial period and during the Holocene. Some landslides are reactivated during wet periods, especially if rainfall is prolonged (Kotarba, 1986). Józefaciuk and Józefaciuk (1995) suggested that about 10% of landslides in mountains and 20% on lowlands are the effect of incorrect engineering practices. In recent years, many old landslides in mountain regions have been reactivated as a result of changes in house building technology. Replacement of wood in favor of heavier construction materials increased ground loading and the risk of landsliding. In July 2001, one of the largest reactivated landslides in the Beskidy mountains covered an area of 15 ha (Bajgier-Kowalska, 2003).

1.9.5.3 Wind Erosion

According to Józefaciuk and Józefaciuk (1995), about 28% of the country is at risk of wind erosion (10% at medium and 1% at high risk). The most at-risk regions are in the central part of the Polish Plain and, to a lesser degree, eastern Poland (Lublin Uplands) and the Sudety and Carpathian foothills. Generally, wind erosion took

place during frosty winters with little snow cover and during sowing periods in early spring and autumn. It is assumed that a wind velocity above 5–8 m s^{-1} is a threshold value to initiate this form of erosion. Wojtanowicz (1990) pointed out that soil particles are usually transported over short distances with average annual losses ranging from 500 to 1000 t km^{-2}, and over longer distances, accounting for about 2–3 t km^{-2}. In extreme cases, transport over short distances of up to 12 000 t km^{-2} was recorded in the Carpathians and in the Lublin Uplands. Gerlach (1966) suggested that the contribution of eolian processes to slope transformation in some parts of the Carpathians is larger than that of water erosion. The effect of the former was assessed as 60% and the latter as 40%. In the east of Poland, the average annual deposition over 8 years was assessed as 100–300 t km^{-2} (Repelewska-Pękalowa and Pękala, 1991). Wind erosion in the Wielkopolska region (Polish Plain) accounted for 500–2000 t km^{-2} (Podsiadłowski, 1994). Most soil removed from fields is deposited along roadside shelter belts. The direct effect of the process was a high spatial variability of soil even within small fields (Stach and Podsiadłowski, 2002). The authors estimated that during seed-bed preparation, wind (pulverizing) erosion reaches on average 580 t km^{-2} on loamy sands and sandy loam of the Polish Plain.

1.9.6 SOIL CONSERVATION MEASURES

After World War II, intensive work to introduce erosion protection measures was started. In most at-risk regions of Poland, demonstration sites were organized to popularize contour farming, contour strip cropping, terracing and special crop rotations (e.g. Ziemnicki, 1955). Scientists were engaged in designing technical structures on stream beds, different methods of protection against gully development (with systems of gully self-filling) and introduction of tree and bush shelter belts. However, with increasing mechanization, the proposed systems became troublesome to maintain and slowly disappeared. Recently, the Ministry of Agriculture and Rural Areas Development and Ministry of Environment (2002) published recommendations for good agricultural practice. They suggest that arable land susceptible to erosion on slopes of >20% should be afforested or turned into grassland and, on slopes of 10–20%, protective measures should be used (anti-erosion rotation with cover). The recommendations are based on guidelines from Józefaciuk and Józefaciuk (1999). All regulations concerning soil conservation at the country level are covered by the Act of Protection of Arable and Forest Land (1995). Another plan for Reconstruction and Modernization of Food Sector and Development of Rural Areas (Ministry of Economy, Labor and Social Politics, 2004) assumes that farm land aggregation will take place. Within such plans, programs for particular regions are being prepared (e.g. Fatyga, 2002).

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1.10

Czech Republic

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1.10.1 INTRODUCTION

The Czech Republic has an area of 78 866 km^2 and belongs to the group of medium-sized European states. It lies between the two chief orographic systems of Europe. The western part of the Czech Republic (the Czech massif) is in the Hercynian system and is known as Bohemia. It has a basin character with the border formed by mountain ranges. The eastern part of the Czech Republic (Moravia) belongs to the Alpine–Himalaya system.

The relief of the Czech Republic consists of 50.1% hills, 33.9% highlands, 11.6% mountains and only 4.5% lowlands. Areas with altitudes below 200 m above sea level form only 5.2% of the whole area. Most parts of the country (78.6%) lie at altitudes of 200–600 m, 38% at 200–400 m and 15.2% at 600–1000 m. Mountainous areas above 1000 m form only 1.1% of the Czech Republic. The highest mountain of the Czech Republic is Sněžka (1602 m), in the Krkonoše Mountains on the border with Poland.

The Czech Republic belongs to the temperate climatic zone with predominantly westerly circulation. The geographic position of the Czech Republic allows the full development of all seasons. Differences in climate are caused by relief and by the increase in continental influences towards the east.

The mountainous areas along the border have the highest precipitation. These are especially Šumava and Krkonoše in Bohemia and Hrubý Jeseník and Moravsko-Slezké Beskydy in Moravia. Here the total amount of

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TABLE 1.10.1 Land cover in the Czech Republic

Land use type	km^2	%
Anthropogenic areas	4455.1	5.3
Arable land	35210.2	42.2
Other agricultural land	10510.3	12.6
Meadows	2310.8	2.8
Woodland	30412.5	36.4
Fenland	83.9	0.1
Water areas	541.2	0.6

Source: CORINE Landcover. Ministry of the Environment of the Czech Republic, 1997.

rainfall is over 1500 mm yr^{-1}. Typical rainfall amounts are between 600 and 800 mm yr^{-1} in hilly areas and highlands. The driest areas of the Czech Republic lie in the north-western part in the rain shadow of Krušné Hory and in the south of Moravia and have amounts between 450 and 500 mm yr^{-1}. On average, 31% of annual precipitation results in surface runoff. The values of the runoff coefficient vary in different regions.

Peak values of runoff usually occur during the spring months. Specific runoff values differ regionally and are determined by the character of rocks, relief, soil, vegetation, etc. The specific runoff values reach 30–40 l s^{-1} km^{-2} in the mountain areas along the border, 5–8 l s^{-1} km^{-2} in the hills and the highlands and 0.1 l s^{-1} km^{-2} in the lowlands.

Land cover in the Czech Republic is shown in Table 1.10.1. The main soil types in the agricultural areas are Cambisols (42.3%) Luvisols (14.6%), Chernozems (14.3%), Illimerized and Gleysols 13.1% and floodplain soils (1.7%).

1.10.2 EROSION PROCESSES AND THEIR HISTORY

Stehlík (1981) mentions several periods of extreme soil erosion intensity. High erosion activity can be influenced by climatic changes, but the most important role in the growth of erosion activity is the development of agriculture. In the prehistorical period, the increase in erosion intensity was slower. The first period of soil erosion increase occurred most probably during the late Bronze Age (around 750 AD). In this period, the population, and with it the area of the arable land, grew substantially. At the end of the Sub Boreal, a drop in temperature occurred, which was accompanied by frequent rainstorms. Sedimentological research shows an increase in floodplain silts by 850 AD. The amount represents half of all erosion sediments from the Neolithic period to the present.

A substantial increase in soil erosion, the consequence of human activity in agriculture, appears in connection with the cold climate of the13th and 14th centuries. This was a period of colonization, in which soil degradation and deforestation of undulating areas assumed great proportions. The evidence was found, e.g., in the floodplain sediments of the Morava River near Uherske Hradiste (Zelnitius and Hrubý, 1939; Demek, 1955).

A further increase in erosion processes occurred between 1750 and 1850 as a consequence of a shift from a three-field system to crop rotation at the end of the Little Ice Age. The reappearance of erosion processes and the substantial growth of gullies in Southern Moravia and in Bohemia are mentioned in the historical literature (Renner, 1934; Láznička, 1957). The effect of historical gully erosion in the form of relic gullies is well documented also by Stehlík (1954), Macka (1955), Gam (1956, 1957), Lochman (1964), Kastner (1981), Buzek (1986) and Kliment (2003). In the late 19th and early 20th centuries, state institutions began to

introduce measures to protect the landscape from gully and river erosion, mainly stabilization with vegetation and check dams.

The natural balance of the landscape was destroyed greatly after 1948, when socialist mass-production ideology was put into practice. The land was consolidated, which brought about new erosion processes. Anti-erosion measures remain insufficient even to the present. The consequence is degradation of soil and a great amount of suspended sediment carried away by waters. The insufficient protection causes extensive pollution of water (Kliment, 1995).

1.10.3 PROCESS OF EROSION AT PRESENT

1.10.3.1 Water Erosion

About 50% of agricultural land in the Czech Republic is endangered by erosion depending on the climatic, morphological and soil situation (Table 1.10.2). At present, water erosion affects approximately 40% of the arable land (Figure 1.10.1).

Mass production in agriculture is considered to be the main cause of water erosion on agricultural land. Mass production was employed between 1950 and 1990 and it consisted of the following elements:

- consolidation of land and establishing large units of arable land (Figure 1.10.2);
- destroying landscape lines and barriers (field roads, ridges, etc.) that prevent surface runoff;
- transformation of grass areas to arable land in sloping areas and in foothills;
- reduction of infiltration capacity of soils by using heavy machinery, which causes compaction;
- using inappropriate methods, especially planting in widely spaced rows;
- lack of appropriate technology for soil-protective cultivation of land.

Figure 1.10.2 shows land-use system changes connected with the 'collectivization process', which accelerated soil erosion processes dramatically in some areas.

The water erosion danger has increased especially in the hilly areas that are used intensively for agriculture, and also in the highlands and foothills of Bohemia and Moravia.

Zachar (1970) mentions severe erosion events in 1960 and 1962; soil loss on arable land reached up to 1000 $m^3\,ha^{-1}$, depending on the crop, soil and slope. Many other cases of local storm events with strong sediment transport and deposition are also documented in the literature.

TABLE 1.10.2 Agricultural land endangered by soil erosion

Water erosion risk	Soil loss ($t\,ha^{-1}\,yr^{-1}$)	Area (ha)	Proportion (%)[a]
Very small	<1.6	134 041	3
Small	1.6–3.0	1 094 507	26
Medium	3.1–4.5	1 054 905	25
Great	4.6–6.0	728 972	17
Very great	6.1–7.5	484 365	11
Extreme	>7.5	782 601	18
Total		4 279 391	100

[a]Of agricultural land.

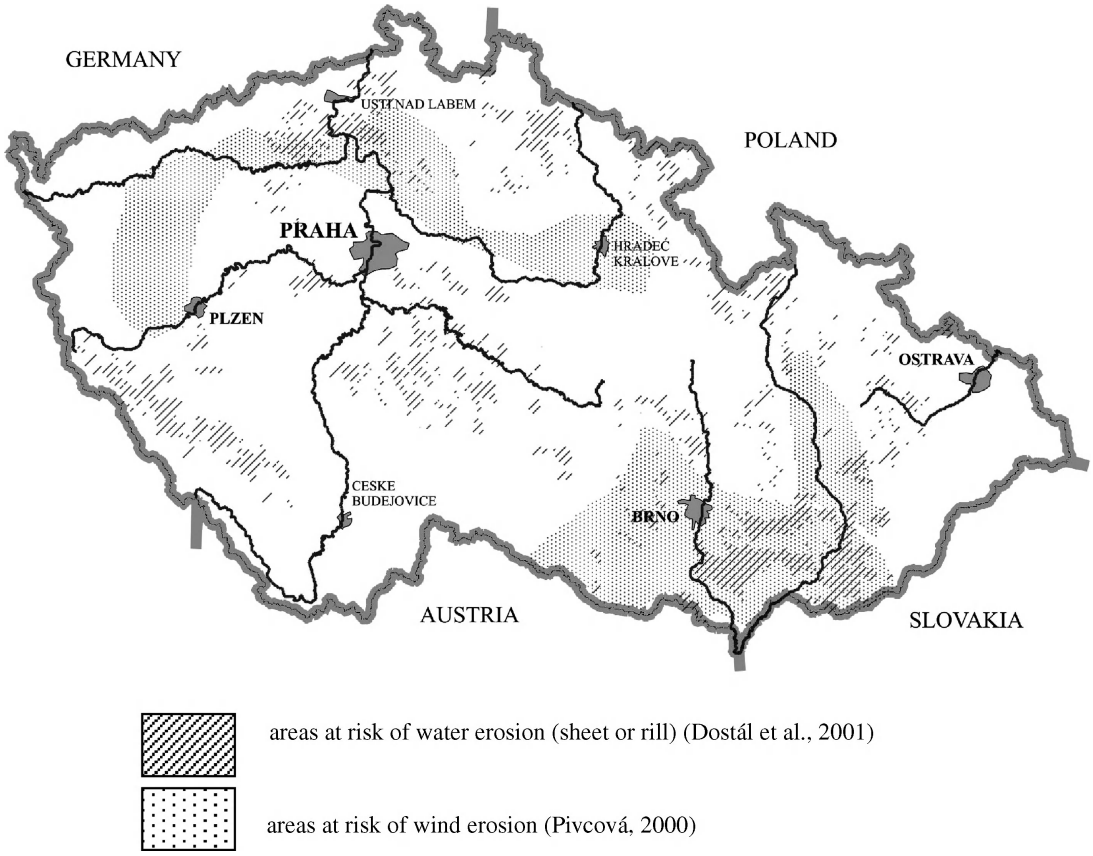
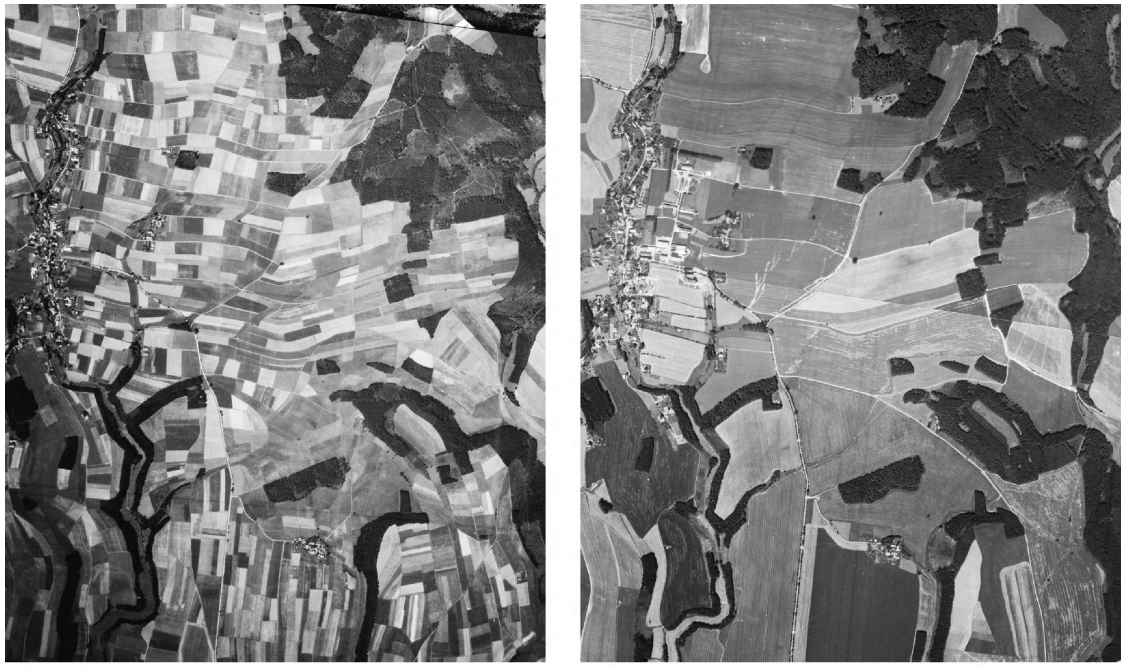


Figure 1.10.1 Areas at risk of water and wind erosion in the Czech Republic

It has been estimated that for the mitigation of erosion risks in the Czech Republic, about 18% of the endangered land can be controlled by crop and vegetation management practices and by soil management (e.g. crop rotation, contouring, conservation tillage, organic matter supply), about 5.7% of the land should be controlled by more efficient conservation measures, such as strip cropping, mechanical control measures (contour bunds, terraces, etc.) and 16.3% of the threatened land requires permanent grass cover to provide adequate protection.

Water erosion endangers the soils in the Czech Republic by decreasing their natural productivity. Also, it influences the retention capacity of the landscape and formation of surface runoff. This is especially of concern in the case of extreme rainfall. Soil protection measures involving system changes in land design and remediation of soils would help to improve the runoff and precipitation regime in the landscape. Also, they would increase the environmental stability of the landscape and its aesthetic value.

In 2001, the map of Erosion Risk and Sediment Transport was published (Dostál *et al*., 2001). It is based on the 1995 data for land use and the USLE method for agricultural land in the Czech Republic. The following crop types were included in the calculation: arable land, orchards, hop-fields and vineyards. The average value of soil loss is $2.27\,\mathrm{t\,ha^{-1}\,yr^{-1}}$. However, the annual R value for the rainfall used was $R = 20.0$ $\mathrm{MJ\,ha^{-1}\,cm\,h^{-1}}$. At present, the value is being revised and the result will probably be a new value close to $R = 50.0$ $\mathrm{MJ\,ha^{-1}\,cm\,h^{-1}}$. The new value would influence the calculation results in a linear way.



(a) (b)

Figure 1.10.2 Land-use system changes. The aerial photograph shows the same location in the foothills of Orlické hory mountains in north-east Bohemia in (a) 1954 and (b) 1984

Tables 1.10.3 and 1.10.4 show the average soil loss depending on slope and altitude. The total annual soil loss on agricultural land within the Czech Republic can be estimated as 9 085 100 t yr^{-1} and the amount of sediment entering water courses as a 3 589 500 t yr^{-1} (Dostál *et al*., 2001).

1.10.3.2 Wind Erosion

Wind erosion endangers approximately 23% of arable land in Bohemia and 40% in Moravia. Wind erosion occurs especially in relatively dry and warm climatic areas with lighter soils. There are several conditions that influence the process of wind erosion. In the case of the Czech Republic, areas needing protection against wind

TABLE 1.10.3 Soil loss on arable land (including orchards, hop-fields and vineyards) depending on slope

Slope (%)	Area (ha)	Area (%)	Average annual soil loss (t ha^{-1} yr^{-1})
<5	2 001 000	56.4	0.76
5–10	1 062 672	29.9	2.92
10–15	348 974	9.8	6.11
15–20	98 898	2.8	9.52
>20	35 338	1.1	13.89
	3 546 882	100	

TABLE 1.10.4 Soil loss on arable land (including orchards, hop-fields and vineyards) depending on altitude

Elevation above sea level (m)	Area (ha)	Area (%)	Average annual soil loss (t ha^{-1} yr^{-1})
<200	219 870	6.1	0.55
200–400	1 653 322	45.8	2.4
400–600	1 479 770	41.0	2.28
600–800	247 950	6.9	2.77
>800	6 893	0.2	4.31
	3 607 805	100	

erosion have frequent winds, precipitation below 550 mm yr^{-1} and light sandy soils and loamy-sand soils (Pivcová, 2000). However, field surveys of wind erosion show its intense occurrence even in areas with heavier clay-loamy soils (Švehlík and Vrána, 1985).

Statistical data document the occurrence of wind erosion in periods with low temperatures and precipitation in spring, when there is not enough vegetation cover. Erosion processes are stimulated by winters without snow – in spring the soil surface becomes dry very quickly (Novák *et al*., 1999).

Wind erosion on agricultural land occurred also before 1950, but rotation of crops and small private plots limited its impact. Erosion processes were considerably stimulated by the intensification of agricultural production in the 1950s. In this period, green cover of the landscape was destroyed and large units of land were created.

At present, projects of land consolidation (master planning) are being introduced. They comprise the return to the original landscape character, decreasing the size of plots, renewing grass cover and natural lines in the landscape. The process is prolonged and expensive.

Systematic surveys concerning wind erosion have been made in the district of Bánov in south-eastern Moravia (Švehlík, 1997). Surveys have been made for 40 years (1957–96). Wind erosion events did not occur in only two years (1958 and 1993). The value of wind erosion was the highest in 1972 – its average intensity was 193 m^3 ha^{-1} yr^{-1}. Volumetric measurements were used to estimate the volume of deposits resulting from wind erosion. The resulting value of erosion intensity should be even higher, because the finest soil particles are transported by wind over much longer distances and could not be included in the measurements.

Preliminary calculations document soil transport in the area. The value in the relevant research area was a loss of 0.4 mm of the plough layer in a year. The loss is 4–5 mm in places with high erosion intensity. At the centre of the dust storm, 2 cm is lost.

1.10.3.3 Erosion at Dumps and in Flysch Areas; Erosion in Areas of Timber Production and of Building Activity

Further types of erosion – erosion at dumps, in areas of building activity, in areas with timber production or with crop growing and harvesting – influence the total amount of erosion in the Czech Republic in a significant way, hence the phenomena, which are rather heterogeneous, are combined. A more detailed survey has not been undertaken in this area yet. Erosion research in the Czech Republic is traditionally focused on the area of water and wind erosion on arable land. Other types are considered less significant, or they have only local and restricted importance. Shallow landsliding is of concern only in the flysch areas in the mountains in the east of the Republic (near to the border with Slovakia), but since the area is predominantly forested, problems are at a local scale and have not been a subject of intensive research. Erosion at dumps is significant in the

north-western areas, where there is intensive production of brown coal. However, these are also only local problems and they mostly concern areas which have not been recultivated (revegetated). The specific morphology of these areas prevents further transport of sediment outside the dump area.

1.10.4 PROBLEMS RELATED TO EROSION PROCESSES

The conditions for the occurrence of erosion in the Czech Republic are fairly specific. During the transition to large-scale agriculture and during intensification of agricultural production, the problem of erosion was underestimated, as were its consequences on productivity and damage to land in towns and cities. Also, the negative effect of erosion on the quality of water and occasional damage by wind erosion mainly in central Bohemia and southern Moravia was overlooked.

Apart from water and wind erosion, snowmelt erosion also occurs in the Czech Republic.

Water erosion mostly affects the land in the foothills of the mountains along the border and in the Czech–Moravian Highlands, mainly because the higher parts are usually covered by forest and other land is less sloping or level.

Erosion deprives the agricultural land of its most valuable part, topsoil, diminishes the quantity of the soil profile, decreases the amount of nutrients and humus, damages crops, makes the movement of agricultural machines difficult and causes loss of seed, fertilizers and pesticides. In the case of wind erosion, mainly vegetables are affected. Degradation of the soil as a consequence of erosion diminishes the productive potential of the soil.

Water erosion has many negative effects on the land (so called off-site effects) (Dostál, 1998). The frequent occurrence of storm events, causing high-rate erosion and sediment transport, and their impact on water-courses, water reservoirs, infrastructure and urban areas emphasize the necessity for control of soil erosion problems. Except for direct damage, the influence on water quality, mainly by phosphorus transport from non-point pollution sources and consequent eutrophication, is one of the most visible effects of long-term uncontrolled erosion processes.

1.10.5 SOIL EROSION MEASURES AND POLICY IN EROSION AND THE CONTROL OF OFF-SITE PROBLEMS

The transformation in agriculture that has been taking place in the Czech Republic since the early 1990s has not so far brought a visible improvement in the field of protection against erosion. The transformed agricultural associations and new agricultural entities usually work on large land units that have been created in the past. The way to improve such a condition is to plan complex land arrangements, in which protection against erosion is an inseparable part of the solution. Also, protection against erosion needs to be supported within the grant programmes of the Ministry of the Agriculture and the Ministry of the Environment. The amendment of the law for land protection, which is currently being prepared, could also contribute to better protection against erosion.

Particular ways of protection are chosen on the basis of their efficiency, the desired decrease in soil loss and the necessary protection of objects with respect to the interests and rights of land owners, the environment and landscape protection. In most cases it is a complex of organizational, agro-technical and technical measures, complementing one another and respecting the current basic requirements and possibilities of agricultural production.

As for the organizational solutions, often it is decided – with the slump in agricultural production – to grass over, and sometimes even to forest, arable land.

In a few cases, the plants that protect land against erosion are grown in the endangered area; sometimes the sloping land is divided into several units with different plant cover.

As for agro-technical solutions, conservation tillage is often chosen, mainly for economic rather than protective reasons. Currently it is estimated that those technologies are applied on one-quarter of the agricultural land in the Czech Republic. The high cost of machines necessary for such technologies is a hindrance to their further expansion.

As for technical solutions, during the period 1960–80, mainly in southern Moravia, terraces were built on 5000 ha aiming to make sloping land accessible for large-scale production, namely for orchards and vineyards.

Diversion ditches and contour bunds have also been built to collect eroded soil and runoff and to limit local flooding.

Apart from the general proclamation of the law concerning the protection of agricultural soil, Czech law lacks directives that in the case of damage caused by erosion could be used to decide whether the damage was caused by breaching of the principles of protection against erosion or whether the cause was the occurrence of an extreme downpour of rain. The main motivation to protect land against erosion is not to prevent damage done to the soil but rather to prevent the damage done to towns and cities. If a particular town is stricken with erosion, usually a project and sometimes soil erosion measures in the basin follow, but prevention is much more desirable and effective.

The neglect over a long period of protection of agricultural land against erosion led to silting up of small ponds by sediment. This causes both quantitative problems (small amount of collected water in the pond, low degree of protection against floods) and qualitative problems (eutrophication and negative consequences for the quality of the water in the ponds).

In the Czech Republic, there are currently around 25 000 small water ponds with a total water content of roughly $420 \times 10^6\,m^3$. An expert survey established that the ponds contain as much as $200 \times 10^6\,m^3$ of sediment, which means that half of the ponds' capacity cannot be effectively used. A field survey in selected ponds estimated that the increase in sediment is around $359\,000\,m^3\,yr^{-1}$ (Generel, 1997), and therefore to maintain the current state it would be necessary to excavate every year precisely this amount of sediment. Similarly, an expert assessment of the thickness of the sediment was undertaken. The volume of sediment was divided into three categories: 1, sediment up to 20 cm; 2, sediment of 20–40 cm; and 3, sediment over 40 cm. The estimated volume of sediment in the first category is around $8.4 \times 10^6\,m^3$, in the second category roughly $114 \times 10^6\,m^3$ and in the third category around $740 \times 10^6\,m^3$. At the same time it was established that the volume of sediment equaling the third category has to be removed immediately and in the second category within the next 7–15 years.

Another important problem is the fact that part of the sediments contains toxic or contaminated materials, which have to be removed to dangerous waste sites. On the basis of those data, the cost of mining and neutralization of sediments in the ponds have been estimated at CZK 30 billion (~€1 billion) (Vrána and Beran, 1998).

The State offers financial help to the owners of the ponds. These programmes are directed by the Ministry of the Environment (Section for Revitalization of River Systems) and by the Ministry of Agriculture of the Czech Republic (financial programme for mining of mud from ponds) and another source is the State Fund for the Environment (created from fines and sanctions imposed for breaching of environmental limits).

The Section for Revitalization of River Systems aims to rectify the consequences of damage to the water regime due to pollution and reduced water quantity in basins of small rivers and streams.

The special programme for sediment excavation from ponds was established by the Ministry of Agriculture and for 2003 has been allocated the sum of CZK 400 million (~€13 million).

Compared with the sum necessary to remove mud from all silted-up ponds in the Czech Republic, however, even the sum of CZK 400 million (~€13 million) is insufficient, because the accumulation of sediments in ponds still continues. Apart from that, the dredging of sediment is not a solution to reservoir silting if source

TABLE 1.10.5 Experimental Localities

Plots

Location	Plot size (m)	Measurement period	Slope (%)	Soil type	Annual rainfall (mm)	Land use	Soil loss ($t\,ha^{-1}\,yr^{-1}$)
Velke Zernoseky	3 plots, 20×6 m	1959–80	44.5	Loamy soil (Rendzina)	500	Bare soil/ grass strips/grass	29.7[a]
Třebsín	9 plots, 35×7 m	Since 04/1986	14	Eutric and Dystric Cambisols	510–580	Arable land	Plots serve for measuring soil loss in different conditions of plant growth. Soil loss from 0 (grass) to 20.0 (arable)

[a]With three extreme erosion events excluded (78.6 $t\,ha^{-1}\,yr^{-1}$, for whole measured period).

Catchments

Location	Area (km^2)	Measurement period	Slope (%)	Soil type	Annual rainfall (mm)	Land use	Soil loss ($t\,ha^{-1}\,yr^{-1}$)
Černičí	1.40	Since 04/1992	7	Cambisols, Stagno-gleyic Cambisols, Hystich Gleysols	650–750	67% arable 14% meadows; 18% forest; 1% others	1.5: average over 10 years, 10.0 in 2002

areas of erosion within the watershed have not been controlled systematically. Furthermore, the land consolidation process that should also consider soil and water conservation planning and implementation of measures has made slow progress in the Czech Republic. Finally, for information, experimental localities are summarized in Table 1.10.5.

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1.11

Slovakia

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1.11.1 NATURAL CONDITIONS

The total area of Slovakia (Figure 1.11.1) is 49 050 km^2. Its northern and central parts belong to the Carpathians and its south-west and south-east parts to the Pannonian Basin. The Carpathians are an arch-like mountain system elongated in a west–east direction. The extensive Pannonian Basin penetrates into the Slovak territory in the form of three separated lowlands, namely the Záhorská nížina Lowland and the Danube Lowland in the south-west and the East Slovakian Lowland in the south-east. The altitudinal range of Slovakia is from 95 to 2655 m in the High Tatras. Lowlands cover 40% and uplands 60% of the Slovak territory. The percentage of the areal extent of uplands consists of 45% of low uplands of 300–800 m, 14% of middle uplands of 800–1500 m and only 1% of high uplands of more than 1500 m. The Carpathians contain numerous marked intramountain basins.

The geological structure of the Slovak territory is very heterogeneous. The northern, outer part of the Carpathians is built of Paleogene flysch rocks (alternations of sandstones and claystones). The central Carpathians consist of the so-called core mountains built of Paleozoic crystalline rocks in their central parts (cores) and complexes of Mesozoic sedimentary rocks, mainly limestones and dolomites in their marginal parts. The southern, inner part of the Carpathians is built of Neogene volcanic rocks. The above rock complexes differ significantly in their resistance, resulting in a regolith of changing thickness. The intramountain basins and especially lowlands in the Pannonian Basin are built of sedimentary Tertiary rocks of the lowest resistance. However, the lowlands are almost completely covered by thick layers of Quaternary deposits, mostly fluvial sandy gravels, loess and blown sands.

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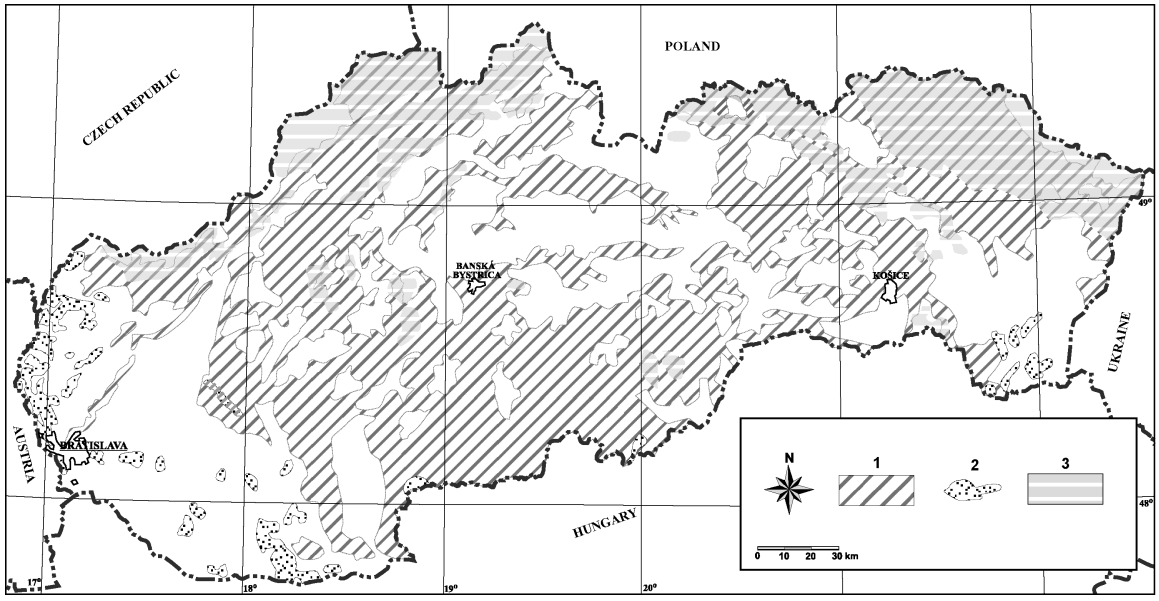

Figure 1.11.1 Map of Slovakia (from Šúri *et al.*, 2002; Klukanová *et al.*, 2002; Liščák, 2002). 1, Water erosion hot spots; 2, wind erosion hot spots; 3, areas affected by or prone to landsliding

Spatially highly variable geological substratum, heterogenous geomorphological and hydrological conditions together with the moderate climate in the contact zone between continental and oceanic influences resulted during the Holocene in the creation of a mosaic of soil types developed mostly under forest and only in some lowland areas of limited extent under steppe. The most widespread of them in the lowlands are Fluvisols, Chernozems and Luvisols, and in the mountains Cambisols, Rendzinas and Podzols. Soil erosion was limited owing to development of a rich vegetation cover and erosion impact was negligible until the Neolithic.

1.11.2 HISTORICAL EVOLUTION OF SOIL EROSION

Since Neolithic times, it has been possible to date the gradual transformation of the original natural landscape into a cultural landscape. The most important human intervention in the past was forest clearance of large areas for the development of agriculture and pasture and extraction of timber for metallurgy. Deforestation was a long-term process. It gradually expanded from the lowlands to the foothills and mountains in relation to individual stages of settlement. One exception was what was called the 'shepherd colonization' when deforestation proceeded in the mountain belts. The result of long-term evolution of farmland was a typical land-use pattern represented by a mosaic of small, narrow plots, tilled both down the steepest slope and along the contour, as we know it from the first half of the 20th century.

After World War II, the original land-use pattern was changed because of the introduction of large-scale mechanized agriculture connected with collectivization. Collectivization resulted in the merging of former small private plots into large cooperative fields, removal of the dense network of artificial linear landscape elements and levelling of terraces created by long-term contour tillage (Figure 1.11.2). The main land-use (land-cover) types in the current landscape are agricultural land (45%, of which 34% is arable land), occurring

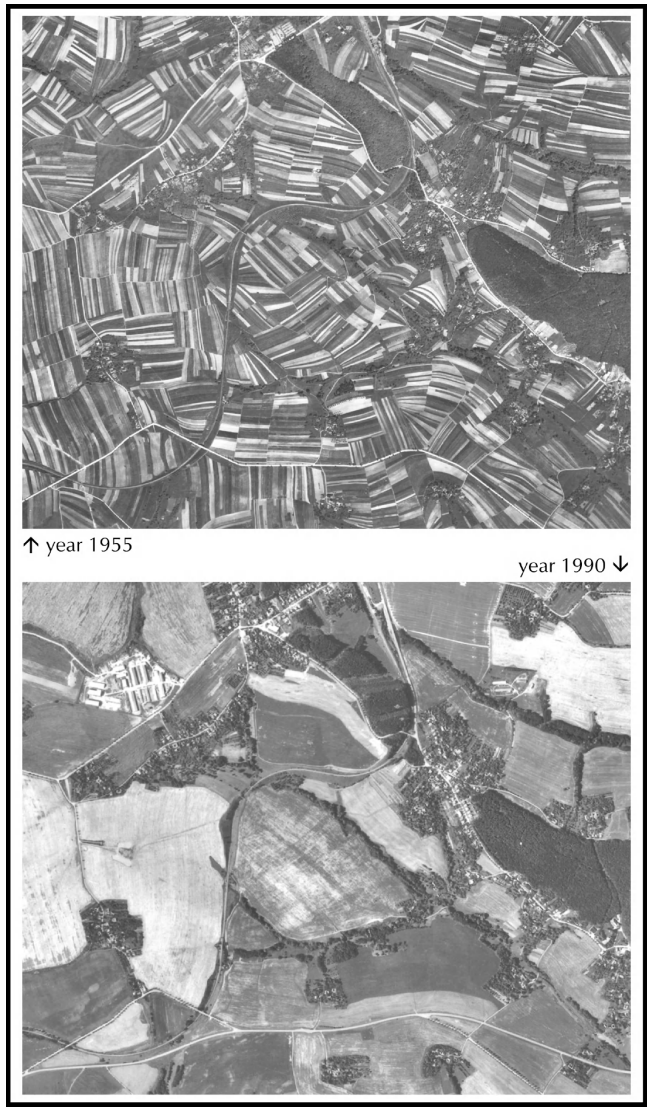

Figure 1.11.2 Example of land use patterns in 1955 and 1990 showing landscape transformation due to collectivization of agriculture (the Myjava Hilly Land; the area in the surroundings of the villages of Poriadie and Rudník). (Reproduced from *Field Excursion Guidebook*, Bratislava, 1999, with permission of the Soil Science and Conservation Research Institute)

predominantly in lowlands and intramountain basins, and forests (38%), distributed mostly in mountains (Feranec and Ot'ahel', 2001).

In general, the Slovak territory is markedly susceptible to erosion processes due to natural conditions, with considerable vertical and horizontal relief dissection. This relatively high potential threat has changed, as a consequence of the historical transformation of the woodland into farmland, into the frequent to regular occurrence of actual erosion processes. Deforestation and the subsequent use of land for grazing and setting up

fields meant such a big change in relations between single components of the natural landscape that it provoked major changes in the course and dynamics of these processes. Water erosion became the dominant, large-scale geomorphic process in deforested areas, which was accelerated owing to intensification of human interference. The best conditions for the effective operation of water erosion are in the agricultural areas lying in wide contact zones of lowlands or intramountain basins with mountains, and also in the intramountain erosional depressions, remarkable for a relatively high share of medium- to low- resistant rocks with a thick cover of easily erodible regolith. The best conditions for intense wind erosion are in selected parts of all three Slovak lowlands.

Gradual transformation of the original natural landscape into the contemporary landscape, together with climatic oscillations in the past, controlled the areal extent, frequency and magnitude of soil erosion events in time and space. Phases of the greatest effectiveness of soil erosion processes were associated with periods of simultaneous occurrence of human interference and higher frequency of extreme rainfall events during colder and wetter climatic fluctuations. The Slovak territory suffered the consequences of four main erosion phases in the past, and at present it is suffering the fifth period of increased erosion. The oldest stage is the Bronze Age. The second stage of increased erosion corresponds to growth of population and subsequent expansion of settlement proceeding from the Danube Lowland to the piedmont areas of the Carpathians and intramountainous basins along the main rivers in the time of the Great Moravian Empire in the 8th and 9th centuries. The third stage of intensification of soil erosion took place in the 13th and 14th centuries, when during what is called the 'great colonization' humans started to settle also the mountains to extract minerals. All three of the above-mentioned stages of inreased erosion were documented by the results of archaeological and sedimentological research on correlated deposits in floodplains of the principal rivers (Bučko, 1980; Stehlík, 1981).

Historically, the fourth stage of markedly increased erosion, characterized by the most distinct geomorphic effect, is linked with the period from the 16th until the 19th centuries. This period of extreme erosion was connected with the combined influence of the Little Ice Age and the 'kopanitse' settlement which originated as a product of the youngest colonization waves (namely Walachian, Goral and kopanitse colonization). Detailed studies of permanent gullies in the territory of the Myjava Hilly Land revealed a clear linkage of these relic gullies to the old, pre-collectivization land use pattern. Gullies were formed mostly along artificial linear landscape features such as access roads, paths, baulks, borders separating the fields, headlands and drainage ditches, with fewer on pasture. The maximum gully density reaches locally up to 11 km km^{-2}, the single gullies are often 10–15 m deep, more rarely up to 20 m and occasionally exceed 20 m. Their formation is a result of two phases of disastrous gullying, the first some time between the second part of the 16th century and the 1730s and the second roughly between the 1780s and 1840s (Stankoviansky, 2003a,b).

The last, i.e. fifth, stage of marked acceleration and increased effectiveness of soil erosion was a response to the introduction of large-scale mechanized agriculture, starting in the middle of the 20th century and lasting to the present. It represents the first period of accelerated soil erosion conditioned exclusively by human interference. A detailed investigation in the Myjava Hilly Land showed that land use pattern adjustments resulted in a change from predominant linear (gully) erosion, typical of the previous stage of accelerated soil erosion, to a prevalence of areal erosion, manifested by a marked spatial increase in areas affected regularly by intense sheet wash, rill and inter-rill erosion. The land use changes influenced also the operation and effectiveness of linear erosion. In contrast to the past, almost exclusively topographically controlled ephemeral gullies were formed in this period. The increased intensity of soil erosion after collectivization is confirmed above all by deposits, commonly reaching thicknesses up to 1 m at footslope positions or even more in the case of fill of some cuts or gullies incised along thalwegs of narrower dry valleys (Stankoviansky, 2003b).

Further evidence for the intensification of soil erosion in this period is the increase in the occurrence of muddy floods. This reflects the considerable increase in geomorphic effectiveness of extreme meteorological–hydrological events under modern conditions, while their frequency is comparable with the pre-collectivization

period. Flashy muddy floods represent a significant environmental issue and natural hazard for local inhabitants (Stankoviansky, 2002).

1.11.3 THE MOST IMPORTANT CURRENT EROSION PROCESSES AND THEIR SPATIAL DISTRIBUTION

Natural conditions, historical evolution of spatial distribution of farmland, its contemporary spatial organization and land-use pattern influence current soil erosion risk. Of the total area of Slovakia, agricultural land occupies about 24 420 km^2 and 57% of it is affected and/or threatened by water erosion. Concerning the arable land, historically concentrated in lowlands, intra-Carpathian basins and partially also in lower portions of mountains, of the total area 14 600 km^2 about 42% is affected and/or threatened. The highest degree of wind erosion threat involves 391 km^2 of sandy and 1712 km^2 of loamy–sandy soils.

The most significant current erosion process, degrading both agricultural and forest soils in Slovakia, is represented by areal and linear water erosion due to surface runoff of rainfall and snowmelt waters acting especially during extreme rainfall events. It occurs above all on farmland, much less on woodland and to a limited extent in positions above the upper timber line.

Naturally, the most effective water erosion affects mostly extensive areas of agricultural land where it can occur on practically all inclined parts of the relief with suitable natural characteristics for its operation. The most dangerous events are in May and June when the soil surface on arable land is unprotected or only weakly protected by vegetation (Figure 1.11.3). The frequency of spring extreme events is locally fairly high; for instance, in the Myjava Hilly Land there were 1–3 extreme spring events yearly in the period 1993–96 (Stankoviansky, 2003b). The greatest effect of rainfall events within the last decade of the 20th century occurred in 1993, 1994, 1996, 1997 and 1999 (Jambor, 1999, 2000). The geomorphic effectiveness of snowmelt events in March (Figure 1.11.4), although fairly high in some years (e.g. in 1993 and 1999), is in fact much lower than in the case of relatively frequently occurring heavy rains.

The most affected intensely agriculturally used areas in the Carpathians are situated especially in the flysch and volcanic belts, namely in lower mountains, in submountainous landscapes in intramountain basins and erosional depressions, and also in wider valleys. Among the most affected geomorphic units belong, for example, the Šarišská vrchovina Mountains in eastern and the Myjava Hilly Land in western Slovakia. The most affected lowland areas are represented above all by higher and more dissected parts of loessic hilly lands, especially in the northern and eastern periphery of the Danube Lowland. The spatial distribution of areas affected by water erosion in Slovakia is faithfully depicted on the map of actual water erosion by Šúri *et al.* (2002) at the scale of 1:500 000. This map is based on the USLE while individual erosion categories are expressed qualitatively. The land was grouped into six erosion classes ranging from negligible to extreme. The spatial distribution of areas classified as heavy, very heavy and extreme erosion was generalized to show hot spots in Figure 1.11.1.

The most important role among water erosion processes is areal erosion, understood as the joint operation of sheet wash, rill and inter-rill erosion. Rill erosion in the form of rills of various size (rarely exceeding depths of 30 cm) and shape is regularly erased by tillage operations following an erosion event and therefore this process is often unnoticed. The hidden character of water erosion processes as a whole helps them to escape from the centre of attention of environmentally oriented research and practically implemented environmental policies. Hence although water erosion has seriously affected much arable land with sloping topography within recent decades, the land users do not take this problem into consideration.

Linear water erosion in current conditions, unlike in the past, is predominantly topographically controlled, giving rise to ephemeral gullies of two different forms: first wide (up to 5–6 m) and shallow (up to 25–30 cm), cut into the cultivation layer. Ephemeral gullies of this type are formed mostly as a consequence of

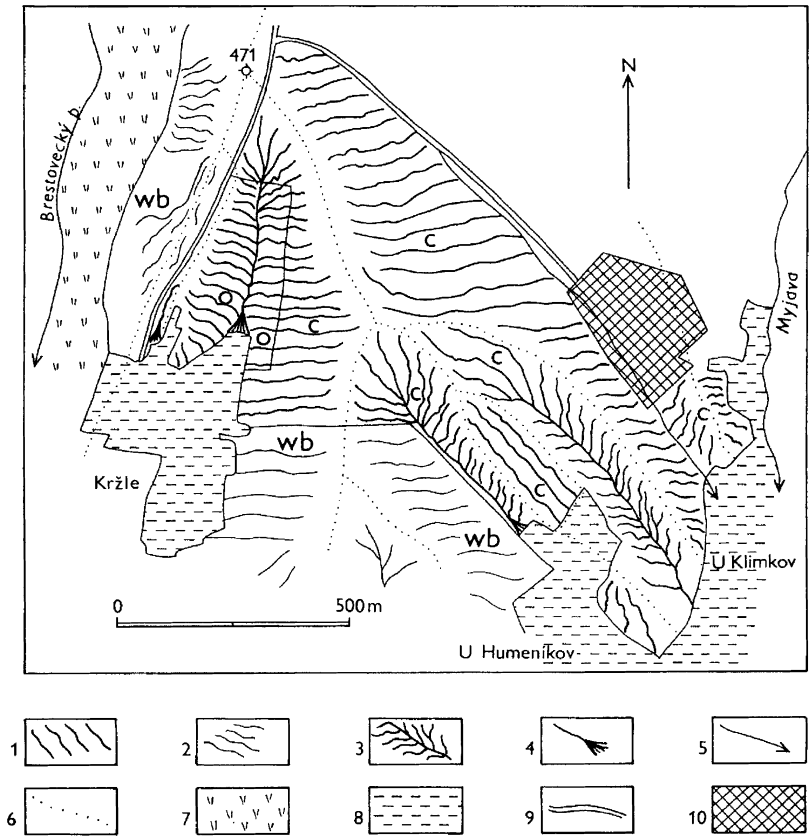

Figure 1.11.3 The geomorphic effect of rill and concentrated flow erosion triggered by heavy rainfall (6 May, 1993; the Myjava Hilly Land, the Upper Myjava Catchment). 1, Marked rills; 2, less expressive rills; 3, ephemeral gullies; 4, colluvial fans; 5, streams; 6, watersheds; 7, meadows; 8, hamlets; 9, roads; 10, area of collective farm; wb, winter barley; o, oats; c, corn. (Reproduced from *Geografický Časopsis* 1997, **49**: 3–4, with permission of the Institute of Geography, Slovak Academy of Sciences)

high-intensity, low-frequency rainstorms. Much more rarely they cut into the compacted plough pan; in such cases, their depth reaches up to 1 m and, exceptionally, more. Ephemeral gullies of this type are formed usually as a consequence of a low-intensity, high-frequency rainstorms. Both forms of ephemeral gullies are erased regularly by conventional tillage or, in exceptional cases, by heavy equipment. The ephemeral gullies after obliteration form again in the same places during the next extreme event (Stankoviansky, 2003b).

In contrast to arable land, fresh gullies on pastures can (in the absence of tillage) survive and grow gradually into permanent gullies (Kňazovický, 1962). However, current gully erosion is not comparable to the disastrous gullying from the times of the Little Ice Age.

Water erosion in forest environments as a result of high anti-erosional effectiveness of forests is very limited. However, this function of forest is in many places markedly weakened, namely by large-scale forest clearance, incorrect skidding technologies and construction of unpaved roads, ski tracks and ski lifts (Midriak, 1988), resulting in serious damage due to both areal and linear erosion. It is evident that the overall erosion in forest areas increased considerably in recent decades as a result of mechanization of timber harvesting.

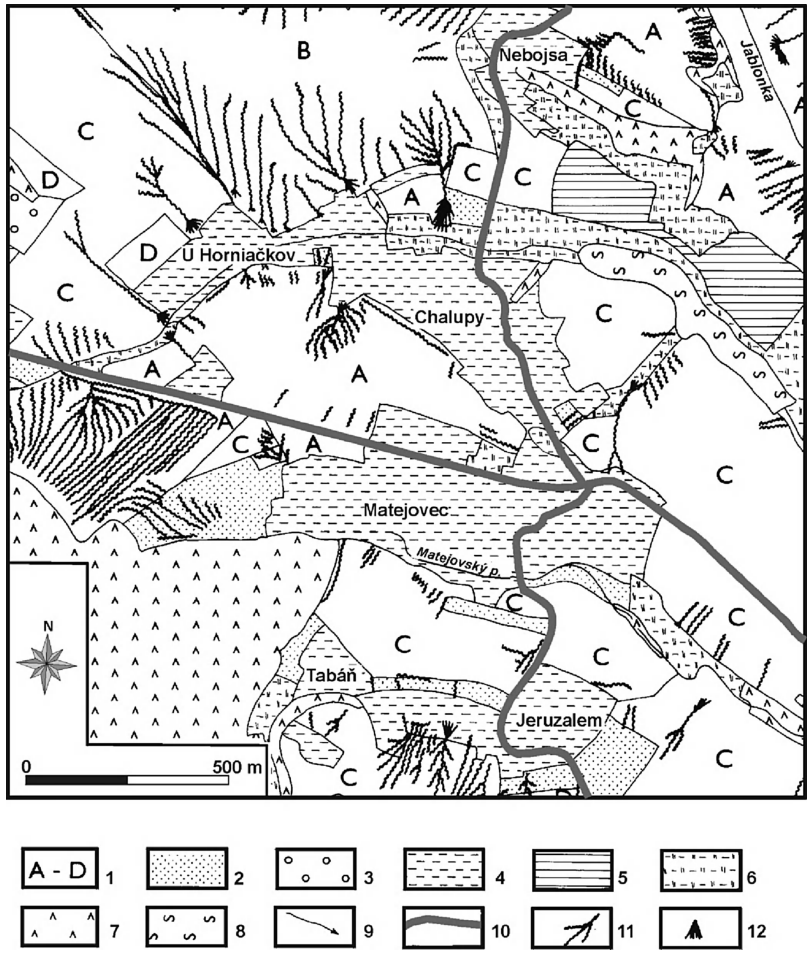

Figure 1.11.4 The geomorphic effect of rill and concentrated flow erosion triggered by snowmelt on different land uses and types of cultivation (March 1993; the Myjava Hilly Land, the Upper Jablonka Catchment). 1, Cooperative fields (A, winter wheat; B, oil-seed rape; C, ploughed fields; D, clover, alfalfa, grass); 2, private parcels; 3, large-scale orchards; 4, hamlets; 5, area of collective farm; 6, meadows and shrubberies; 7, forests and belts of trees; 8, water reservoirs; 9, streams; 10, roads; 11, rills and ephemeral gullies; 12, colluvial fans. (Reproduced from *Proceedings 'Vybrané problémy súčasnej geografie a príbuzných disciplín'*, Faculty of Natural Sciences, Comenius University, Bratislava, 1995, p. 88, with permission from Comenius University)

Relic gullies, formed in the past in agricultural land and now lying under forest, are also not totally inactive. Especially afforested gullies situated on lower slope portions below fields on the upper slope parts (where the runoff is concentrating) are fairly active during extreme events, although the effectiveness of erosion is much lower than during the time of the formation of these gullies.

Water erosion processes above the upper timber line are neglible but here and there their intensity reaches high values.

Wind erosion represents another serious environmental threat, although the areal extent of its operation is much smaller than in the case of water erosion (Figure 1.11.1). Lowland areas with conditions of frequent

moisture deficit, especially with soils that are light in texture, are regularly affected by wind erosion. The major area with sandy soils is situated in the Záhorská nížina Lowland in western Slovakia, covered mostly by aeolian sands. Those parts of the sand dune areas that were not afforested by secondary pine forest, preventing deflation, and are still used as arable land, often suffer from sandy storms. Other smaller sandy areas very prone to wind erosion are scattered in some parts of the Danube and East Slovakian Lowlands. Wind erosion also affects, to a certain extent, selected portions of the southernmost belt of Central Slovakia. Apart from major hot spots with sandy soils, wind erosion affects also areas with loamy soils both in flat and hilly parts of lowlands. Of course, the wind erosion intensity on loamy soils is much lower, but in dry summer periods after the crops have been harvested dust storms can be observed occasionally. According to Stred'anský (1993), the highest intensity of wind erosion occurs in the winter and early spring months when the most favourable conditions for effective wind operation occur, i.e. a frozen, dusty soil surface without vegetation cover. Under such conditions, wind erosion starts at a wind velocity of 4–6 m s^{-1}. Wind erosion is encouraged by the size and shape of contemporary fields, reaching in the most affected lowland parts 100–200 ha, and exceptionally even more.

The first attempt to point out the role of tillage in soil degradation and landform transformation dates back to the 1950s (Lobotka, 1955, 1958). Stankoviansky (2001) studied the long-term geomorphic effect of the combination of irregularly acting water erosion and regular tillage erosion in the Myjava Hilly Land. The geomorphic effect is represented by the lowering of the surface of slopes and ridges in portions ploughed along the gradient and by the creation of cultivation terraces (steps of terraced fields) in portions ploughed along contours (however, these steps were mostly levelled in the period of collectivization). The main role in terms of landform transformation was played by tillage erosion, which was the decisive areally acting geomorphic process in the arable landscape. The estimated thickness of the removed layer in the arable land of the Myjava Hilly Land within the whole cultural period often locally markedly exceeds 1 m (Stankoviansky, 2003b).

1.11.4 RATES AND EXTENT OF EROSION PROCESSES

The first attempts to quantify soil erosion rates in Slovakia date back to the 1950s and 1960s when volumetric methods were occasionally used to calculate soil loss by rill and gully erosion. Later, in the 1980s and 1990s, a wide range of the methods for the determination of soil erosion rates were used, especially plot studies (small monitoring plots with collection of runoff and sediment and medium-sized monitoring plots with tipping buckets); suspended sediment load measurements in zero-order catchments using the Parshal flumes; investigations using the ^{137}Cs method; the measurement of accumulation of eroded material on footslopes, dry valley bottoms and in thalweg cuts using both buried objects and soils; the measurement of sedimentation in channels of local streams using dendrochronology; the measurements of siltation in small reservoirs on streams by volumetric methods (Table 1.11.1); and the sampling of sediment load in larger rivers and measurements of sedimentation in large reservoirs.

The results obtained by several researchers using the volumetric method in the 1950s and 1960s to estimate the volume of rills and ephemeral gullies formed during particular extreme rainfall events were summarized by Zachar (1970). One of the most spectacular events was reported by Lobotka (1955) from the early 1950s in the Čekovce in the Krupinská planina Mountains. From his data on the number and size of rills and area of the field, the approximate erosion rate was estimated at 560 t ha^{-1}.

The first studies on small plots were carried out by Stašík *et al.* (1983). The only site with a plot of 25 × 2 m was situated on a slope of 6–7° near Čečejovce in the Košice Basin, Eastern Slovakia. The measurements were made during growing seasons in the period 1981–82 and the mean gross soil loss of these two growing periods was 4.8 t ha^{-1} (with a maximum of 6.8 t ha^{-1}). Later these measurements were repeated in the period 1986–88 at Stakčín and the Ubl'a sites in the Beskydské predhorie Foothills (eight plot/year data in total) using a similar

TABLE 1.11.1 Review of the most important results of measurements of water erosion rates in Slovakia

Site and authors	Method	Size of the site	Process represented	Period	Topography			Vegetation[a]	Mean erosion rate		Range of erosion rates	No. of measurements
					Land form	Length (m)	Inclination (°)		original units	t ha^{-1}	t ha^{-1}	
Čekovce (Lobotka 1955)	Volumetric	93 × 45 m (4185 m^2), 69 rills[b]	Extreme rill erosion	1 extreme rainfall, end of summer, 1st half of the 1950s	Slope	93	18	PF	470 m^3 ha^{-1}	560[c]	560[c]	1
Čečejovce (Stašík *et al.* 1983)	Plot study; total collection	25 × 2 m (50 m^2)	Sheet and initial rill erosion	1981–82 (growing seasons)	Slope	25	6–7	OR, WW	485 g m^{-2}	4.85	2.9–6.8	2
Stakčín, Ubl'a (Chomaničová 1988)	Plot study; total collection	10 × 5 m (50 m^2)[d]	Sheet and initial rill erosion	1986–88 (growing seasons)	Slope	10[d]	6-10	WW, WR, SM, P	294 g m^{-2}	2.94	0–8.7	8
Osikov, Kočín, Gbely, Smolinské, Rišňovce (Fulajtár and Janský 2001)	Plot study; total collection	20 × 2 m (40 m^2)	Sheet and initial rill erosion	1994–96 (whole years)[e]	Slope	20	4-6	SB, SF, GM, WW, ShB	925 g m^{-2}	9.25	0–75[f]	12
							8–10	WW, SuB, WR, SB, GM, SF, AL, OR	1384 g m^{-2}	13.84	0–75	34
Lukáčovce, Turá Lúka (Gajdová *et al.* 1999)	Plot study; tipping buckets	100 × 10 m (1000 m^2)	Sheet and mature rill erosion	1997–99 (whole years)	Slope	100	4–12	WW, WR, SB, OR, GM	42.4 kg ha^{-1}	0.04	0–0.32	10
	Hydrological method: Parshal flume, sediment concentration	elementary watersheds, Luk.: 143 ha T.L.: 77 ha	Overall off-site sediment transport	1997–99 (whole years)	Bed of seasonal stream	400–700	3–10 (max. 14)	WW, SB, SM, GM, P, SF, SB, G, OR	32.3 kg ha^{-1}	0.03	0–0.08[f]	5

(*Continued*)

TABLE 1.11.1 (*Continued*)

Site and authors	Method	Size of the site	Process represented	Period	Topography			Vegetation[a]	Mean erosion rate		Range of erosion rates	No. of measurements
					Land form	Length (m)	Inclination (°)		original units	t ha^{-1}	t ha^{-1}	
Jaslovské Bohunice (Fulajtár 2002b)	^{137}Cs method; Walling's conversion models	Elementary watershed 34.4 km^2	Overall on-site soil redistribution	ca 1954–98	Plateaus, slopes, valley bottom	20–100	4–8	ArL	17.3 t ha^{-1} (26.1 t ha^{-1})[h]	17.3 (26.1)[g]	0–ca 50	70/40/16[h]
Slovakia (Janský 1992)	Volumetric; sediment thickness	Watersheds 0.8–28 km^2	Overall off-site sediment transport	ca 1950–85	Bed of reservoirs	Several km	Variable	F + AgL	2897 m^3 km^{-2}	34.8[c]	2.3–90.6[c]	27

[a]F, forest; AgL, agricultural land; G, grassland; ArL - arable land; AL - alfalfa; WW - winter wheet; WR - winter rye; SB - spring barley; OR - oil-seed rape; P - peas; SM - silage maize; GM - grain maize; SF - sunflower; SuB - sugar beet; P - potatoes; PF - ploughed fallow.
[b]Number of measured rill profiles is not known.
[c]Presuming a bulk density of 0.2 g cm^{-3}.
[d]Not verified.
[e]Except for some short periods during winter and during agrotechnical works (ploughing, seeding, harvest); differs from site to site.
[f]Estimation; capacity of collecting device exceeded.
[g]Mean of all slope positions/mean of transect maximum values.
[h]All sampled points/slopes affected by erosion/points with maximum erosion within each slope transect.

approach by Chomaničová (1988). The mean gross soil loss in this period was 2.9 t ha^{-1} (with a maximum of 8.7 t ha^{-1}).

The largest set of data acquired by the plot studies was provided by Fulajtár (cf. Fulajtár and Janský, 2001; Fulajtár, 2002a). These data were obtained from plots with a size of 20×2 m at five different erosion monitoring stations representing various landscape types (Rišňovce, Kočín, Smolinské and Gbely in western and Osikov in eastern Slovakia) in the period 1994–96. The number of the plots varied during particular years and in total 28 different plots (often twinned – crop and fallow) were involved in the investigation. Some of them existed for only 1 year, others were installed for 2 or 3 years. The gradient of these plots ranged from 3 to 10°. On the plots a range of major agricultural crops were grown, representing the crops of surrounding areas (winter wheat, spring barley, winter rye, oil-seed rape, alfalfa, maize, sunflower and sugar beet). During the 3 years of erosion monitoring, 649 rainfall events were recorded and as sometimes there were several plots on one monitoring station these rainfall events represents 1693 combined plot/rainfall events. From these rainfall events, when soil loss potentially could occur, only 242 erosion events were registered. This represents a 14% probability of the occurrence of soil loss during rainfall events. Among these 242 erosion events, only very few were significant. The 13 greatest erosion events among the total number of 242 account for 50% of the total soil lost from the plots during the whole monitoring period and 66 for 90% of soil loss. This means that only a small number of really significant soil-loss events occur in the agricultural areas and the major part of soil loss is a result of an exceptional coincidence of conditions.

The total registered number of yearly gross soil losses per plot was 77, but excluding the twinned plots and those with black fallow, the total number of gross soil losses is 33 for 8–10° slopes and 12 for 4–6° slopes. The values range from zero to several tens of t ha^{-1} yr^{-1}, but in most cases there was only negligible or no soil loss. Unfortunately, the highest value of 75 t ha^{-1} yr^{-1} is only a rough estimation, because the capacity of the collecting device was exceeded. The erosion rates from the fallow were removed from the data set as they do not represent normal everyday conditions and the remaining data were separated into two groups according to gradient (4–6 and 8–10°). The mean soil erosion rate for the group with steeper slopes was 13.84 t ha^{-1} y^{-1}. This group was divided into subgroups with densely seeded crop and with root crops. The mean erosion rate for the root crops was 24.05 t ha^{-1}y^{-1} and for the dense crops 2.91 t ha^{-1} y^{-1}. The great majority of total soil loss was from plots with root crops (89%).

The distribution of soil erosion during the year shows a distinct maximum in spring. However, differences were observed for different crops. Under densely seeded crops (cereals, alfalfa and oil-seed rape) the maximum was broadly distributed from March to June, whereas for root crops (maize, sunflower and sugar beet) the maximum was much sharper from May to July. However, the total yearly distribution for all crops was almost identical with that of root crops, as the soil loss under densely seeded crops is so small that it has minimum impact on total distribution.

Gajdová *et al.* (1999) investigated the impact of agriculture on the quality of water flows in two zero-order catchments in Lukáčovce (dry loessic hilly land in the northern periphery of the Danube Lowland with intensive agriculture) and in Turá Lúka (submountainous moist flysch hilly land at the foot of the Carpathians with modest agricultural exploitation). On arable land, measurements on medium-sized plots of 100×10 m with tipping buckets were established and off-site effects were measured at the outlet of the catchment. The Parshal flumes were built in small seasonal streams draining both catchments and the suspended load was measured using automatic sediment samplers from 1997 to 1999. In total 10 plot/year data on gross erosion rate on the slopes and 5 years' sediment load data were obtained. The mean gross yearly values are small both for plots and streams. On slopes they range from 0 to 0.32 t ha^{-1} with an average value of 0.04 t ha^{-1} and in the streams they fluctuate from 0 to 0.08 t ha^{-1} with an average value of 0.03 t ha^{-1}. The gross sediment load in the stream in Turá Lúka was somewhat greater, because the flume was exceeded during two short extreme rain events when flooding occurred. The results show a considerable spatial variability in soil loss both between the two sites and within the catchments. In the Lukáčovce catchment with a dryer climate and gentle

slopes (4–6°, maximum 10°), practically no soil loss was recorded and the runoff was very limited, whereas in the Turá Lúka catchment, having a rather moist climate and steeper slopes (3–10°, maximum 14°), the soil loss, although low, occurs commonly (Table 1.11.1).

The ^{137}Cs method was used at several sites in western Slovakia. First, individual transects were investigated at Horné Sŕnie, Luborča, Pata, Voderady, Bzince pod Javorinou and Kostolné, and at all study sites the erosional lowering of slopes since the 1950s did not exceed 5 cm (Lehotský and Stankoviansky, 1992; Linkeš *et al.*, 1992; Lehotský *et al.*, 1993; Lehotský, 1999a). Later, the small catchment at Jaslovské Bohunice in the loessy territory of the Trnava Hilly Land was sampled by a multiple transect approach (Fulajtár, 2000, 2002b). The catchment had 60–100 m long slopes with an inclination of 4–8°. The area sampled was approximately 34 ha and the slopes where the erosion could be presumed occupied 24 ha. The sampling transects were placed in a set of parallel lines oriented down the slope. The total number of sampled points was 70, among which 40 points represent the slope positions affected by soil erosion. Usually in each transect the upper convex slope, middle straight slope and lower concave slope were sampled, but in some transects the number of sampled points on the slope differed. The measured weight concentration of ^{137}Cs (Bq kg^{-1}) was converted to a ^{137}Cs inventory (Bq m^{-2}) and the soil erosion/deposition rate was determined by conversion models developed by Walling and He (1997) – the proportional model (PM), the simplified mass balance model (MBM1) and the standard mass balance model (MBM2). For further interpretation the results obtained by MBM2 were used, as this model takes into consideration the most comprehensive set of parameters. The mean erosion rate obtained by this model is 17.3 t ha^{-1} yr^{-1} (considering 40 sampled points situated on slopes). For PM this value is 22.4 t ha^{-1} yr^{-1}. A representative value of soil erosion would be the average of the maximum rate within each of 16 slope transects. This value is 26.1 t ha^{-1} yr^{-1} for MBM2 and 31.4 t ha^{-1} yr^{-1} for PM.

The erosion rates acquired in the Mochovce area in the loessic territory of the Hron Hilly Land, using the same method, fluctuate in the range similar to that of Jaslovské Bohunice site (Van der Perk *et al.*, 2002).

The soil erosion rates were also studied in woodland of mountain areas, using plots 0.5 m wide and 1.5–6 m long with modified Gerlach troughs used for collection of runoff and eroded soil (Midriak, 1986). Plot studies were used at many sites in different geographical conditions, mostly spruce and beech ecosystems and partially also in fir, pine, larch, oak and hornbeam ecosystems (Midriak, 1993). Absolute and relative soil losses were distinguished with the former representing the actual removal of material to the stream network and the latter its redistribution on the slope. Absolute soil losses in woodland are very low, reaching 18 kg ha^{-1} yr^{-1} on average in coniferous forest and 24 kg ha^{-1} yr^{-1} in deciduous forest. However, these values fluctuate in particular localities, ranging from 1 to 61 kg ha^{-1} yr^{-1}. The lowest absolute soil losses are typical of fir and spruce forests, the average losses are in beech forest and above-average losses in oak and pine forests. The relative soil losses are represented by the redistribution of both the inorganic and organic material, while the displacement of inorganic matter is more significant. The annual redistribution of inorganic material extends from 8 to 891 kg ha^{-1} yr^{-1} (109–323 kg ha^{-1} yr^{-1} on average), depending on the specific conditions of a particular stand. The amount of redistributed organic particles ranges from 15 to 544 kg ha^{-1} yr^{-1} (132–255 kg ha^{-1} yr^{-1} on average).

In general, the results show the high anti-erosional effectiveness of forests. However, the situation is very different in forests affected by human intervention. The influence of anthropogenic activities on water erosion in woodlands was studied in the locality of Komárnik situated in the Laborecká vrchovina Mountains, on flysch rocks, covered by mixed fir–beech forest, and in the locality of Biely Váh in the part of the Kozie chrbty Mountains, on carbonate rocks, covered by spruce forest (Midriak, 1989, 1994). In the first case, the absolute soil losses are 13–20 kg ha^{-1} yr^{-1} and the relative losses are 333–1000 kg ha^{-1} yr^{-1}, and in the second case the values are 22–51 and 423–688 kg ha^{-1} yr^{-1}, respectively. The highest values in both localities were associated with clear-cut areas (Midriak, 1989).

Measurements of the rate of water erosion were made also in the East Carpathian Biosphere Reserve in the Bukovské vrchy Mountains, on flysch rocks (Midriak, 1995a,b). The measured values of the absolute losses by

areal water erosion are relatively low (16–81 kg ha^{-1} yr^{-1}), but the total losses (i.e. absolute plus relative losses) on clear-cut areas approach 1470 kg ha^{-1} yr^{-1}. Much higher soil losses are initiated by linear erosion controlled by tractor wheel tracks, logging tracks, forest roads, etc. Such soil losses are almost 100 times higher than the losses induced by areal erosion (4.4–14 m^3 ha^{-1} yr^{-1}). The average removal from a 1 m length of unpaved skidding roads due to both mechanical scraping of the road surface by logged trees and the successive operation of concentrated flow erosion ranges from 0.13 to 0.61 m^3 yr^{-1}. These measurements indicate that because of the relatively dense networks of unpaved roads in some forested mountains, the erosion rates in such areas may be higher than usually expected.

The result of long-term field work in the areas above the upper timber line based on the use of micro-levelling, stereo-photogrammetry, plot studies with the help of modified Gerlach troughs and volumetric methods is an assessment of the rate of present-day geomorphic processes (including water erosion), expressed by values for slope surface lowering. The calculated values of surface lowering in the individual mountains range from 0.10 to 0.72 mm yr^{-1} (average 0.27 mm yr^{-1}) (Midriak, 1983). However, removal leading to slope lowering is not area-wide but is concentrated on the bare or degraded slope portions (approximately 8% of the total area). Average values of surface lowering due to operation of water erosion represent 0.001–0.007 mm yr^{-1} in dwarf pine and grassland stands and 3.4 mm yr^{-1} on bare surfaces, while the maximum approachs 10 times the average value.

Other data which can be used for the indirect evaluation of soil erosion rates are results of the estimation of colluvial bodies formed by muddy floods, of measurements of sediments in channels of local streams, in small reservoirs, and some data for suspended sediment load in larger rivers and siltation of large reservoirs. All these data are considered in the next section on the off-site effects of soil erosion.

The above overview of data on erosion rates in Slovakia shows promising research achievements. The data were gained by several different methods in different periods and under different geographical conditions, which allowed comparison and verification. Nevertheless, it is evident that for generalization with respect to the whole territory of the country with such diversified natural conditions as in Slovakia more comprehensive sets of data would be needed. Therefore, the overall picture of soil erosion activity and distribution which can be based on available information is only a rough sketch.

All collected data are summarized in Table 1.11.1. The most abundant are data gained on small monitoring plots (56 plot/year data), which are distributed in a wide range of geographical conditions. Most of these sites were established on slopes of 4–10°, which are typical of agriculturally utilized hilly and submountainous areas, and the measurements were from all major agricultural crops. The values from these plots fluctuate from zero to a few tens of tons per hectare and average around 10 t ha^{-1}. It should be kept in the mind that the length of the plots is only 20–25 m and erosion on natural slopes is in fact greater.

Values similar to those from small plots were obtained from the few measurements on medium-sized plots (10 plot/year data). These plots have longer slopes and hence better express the natural conditions. The erosion rates are much smaller than values from small plots (0.04 compared with around 10 t ha^{-1}). This is mainly because medium-sized plots represent mostly densely seeded crops, but even if compared with the mean soil loss of small plots with cereals they are considerably smaller (0.04 compared with 0.78 t ha^{-1}).

The mean soil loss in two zero-order catchments was very small (0.03 t ha^{-1}). This was not very surprising at the drier Lukačovce site but it is more surprising with regard to Turá Lúka where the mean soil loss was expected to be much higher than the recorded 0.08 t ha^{-1}. The rainfall and runoff were rather intensive during the measuring period, two short floods occurred and also rill erosion was observed. It is probable that the relatively high soil resistance controls erosion at this site.

The main disadvantage of all measurements on plots and in catchments is time. The 2–3-year measuring periods are not sufficient to record rare events. This disadvantage was overcome by using the ^{137}Cs method. The main problem of this method is to achieve proper calibration allowing correct conversion of measured ^{137}Cs inventories to soil loss. Nevertheless, the recently used calibration models are providing fairly realistic

values and therefore the results obtained by the ^{137}Cs method can make a great contribution to further erosion rate investigations.

The mean erosion rates for Jaslovské Bohunice were compared with the soil erosion rates obtained for small plots by Fulajtár and Janský (2001). The results provided by the ^{137}Cs method are considerably higher than the data from small plots (approximately 10 t ha^{-1} yr^{-1} for small plots compared with 17 t ha^{-1} yr^{-1} for all slopes in Jaslovské Bohunice or 26 t ha^{-1} yr^{-1} for the most intensively affected parts of the slope transects in Jaslovské Bohunice). However, despite this difference, both datasets fit fairly well considering that they reflect different erosion processes and also slightly different geographical conditions. The small plot data reflect solely water erosion on the slope (the soil loss on the lower margin of the plot), whereas the data obtained by the ^{137}Cs method reflect all soil redistribution processes on the slope (the final balance of erosion and deposition by water, wind and tillage) and the values are available for several points along the slope. The values from the small plots represent short-term erosion rates (1994–96), whereas the ^{137}Cs method provides the mean erosion rates for the period since the mid-1950s. The small plots had standard slope parameters of 8–10° inclination and 20 m length whereas the ^{137}Cs method was applied to slopes of 3–7° inclination and 50–80 m length. The soil and climatic conditions were similar. Considering that the erosion rates obtained by the ^{137}Cs method reflect a more complex set of erosion processes on longer slopes, it is logical that they are higher.

From all that was measured and observed in the field, it can be concluded that the mean soil erosion rates in the agriculturally intensively utilized hilly areas where slope inclinations reach not more than 6–10° and the slope lengths do not exceed a few tens of metres or a maximum of 100–200 m, the mean soil erosion rate can fluctuate around 20 t ha^{-1} yr^{-1}. The thick young to fresh depositional bodies often observed in the field indicate that in submountainous areas erosion rates can be higher, locally maybe markedly, and in some hot spot areas they can reach several tons per hectare. As a confirmation of such a supposition, the erosion effect of a catastrophic storm event, recorded in the early 1950s close to the village of Čekovce, can serve, with the estimation of an approximate erosion rate of 560 t ha^{-1} (see above). Unfortunately, not much is known so far about the frequency of such events.

1.11.5 MAJOR ON- AND OFF-SITE PROBLEMS AND COSTS

1.11.5.1 On-site Effects

Soil erosion has an important impact on the properties of affected land and soil and it causes direct damage to crops. Direct financial losses are caused by loss of nutrients and especially fertilizers applied to soil, and also the removal of major soil components such as humus, which is extremely important for the storage of water and nutrients. The removal of topsoil lowers soil fertility and reduces yields.

The on-site effects of erosion were studied especially with respect to (1) loss of nutrients, (2) changes in basic soil properties after the removal of top soil, (3) territorial extension of strongly eroded soils and (4) impact of reduced fertility on yield of major agricultural crops. Direct damage to crop growth caused by runoff and sediments was not studied, but numerous such events as excavating of crop roots by eroding waters on the slopes where rills are formed and burying of young crop growth by sediments deposited at the footslope were observed in the field.

Loss of nutrients was investigated by Stašík *et al.* (1983) in flysch areas of Eastern Slovakia and by Fulajtár and Janský (2001) in loessic areas of Western Slovakia. Both regions belong among the major hot spot erosion areas of Slovakia. The measured loss of nutrients was not too high (0–0.6 kg ha^{-1} of phosphorus, 0–12.5 kg ha^{-1} of potasium and 0.1–0.7 kg ha^{-1} of nitrogen). The main portion of nutrients was moved in suspension and sedimented along the lower boundary of the field, except for nitrogen which was shared

between suspension and solution. The dissolved part of nitrogen represents the major problem because it is acting as a polluting agent in surface waters and reservoirs. However, nitrogen circulation is related not only to soil erosion, but also to runoff. A special problem which was also investigated together with the loss of nutrients was the loss of organic matter. The measured values ranged from 48 to 390 kg ha^{-1}. Comparing the mean store of organic matter in the soils at the investigated sites, the measured yearly losses represent 0.09–0.76%.

The reduction of soil fertility resulting from long-term soil erosion is documented in a large amount of analytical data from soil surveys done in the 1960s. Most of these data were not published and can be found in the archive of Soil Science and Conservation Research Institute in Bratislava. Later, a few studies were focused directly on changes to soil properties due to erosion. One example from the loessic areas was provided by Fulajtár and Janský (2001). The changes in a whole range of soil characteristics such as organic matter content and quality, pH, cation-exchange capacity, nutrient content and physical properties of selected Chernozem was documented. These changes were expressed also by the change in soil morphology and horizonation. The original 60-cm thick mollic A horizon and 20-cm thick weathered B horizon were removed and replaced by a 30-cm thick ochric A horizon which was formed in the upper part of C horizon mixed with remaining material of A and B horizons by tillage. In this way the original Chernozems were transformed to Regosols.

Much attention was devoted to investigating the extent of strongly eroded soils. The mapping is based on the colour contrast between eroded and noneroded soils. It was successful especially in loessic areas, where the Regosols formed by erosion are identifiable as bright areas surrounded by dark, noneroded Chernozems. Hence the eroded soils can be distinguished on aerial photos and satellite images and in the field. Nevertheless, up to now only a few hot spot areas have been mapped at medium or detailed scales, all of them mostly in loessic areas, namely in the surroundings of Rišňovce village in Nitra Hilly Land and in Levice district (Fulajtár, 1994, 1998, 2002c; Fulajtár and Janský 2001) and in Trnava Hilly Land (Šúri and Hofierka, 1994; Šúri and Lehotský, 1995; Sviček, 2000).

The impact of reduced soil fertility on the yields of agricultural crops can be easily observed in the field. The density and height of growing crops are usually much lower on convex slopes with strongly eroded soils. Another good field indication is the difference in germination of crops and also weeds which usually germinate after the harvest. In less fertile eroded soils, the germination begins much later. Data on such phenomena and key market crops were provided by Fulajtár and Janský (2001). The yields on strongly eroded soils were reduced in comparison with noneroded soils to 76% for winter wheat, 35% for spring barley, 65% for grain maize and 58% for sunflowers.

1.11.5.2 Off-site Effects

Eroded material both from farmland and woodland is carried away. The most important sediment transfer is represented by muddy floods. Most of the sediment coming from fields is deposited in local positions on footslopes and in valley bottoms close to them and the rest travels into the streams and further to reservoirs.

The geomorphic effects of muddy floods in the form of muddy deposits in positions beyond the fields have been noted by researchers dealing with soil erosion in various parts of Slovakia since the 1960s. They documented the consequences of single, isolated extreme rainfall events, while the selection of study localities was influenced exclusively by the place which was affected by a particular event.

More systematic, repeated observations of the effects of muddy floods started to be conducted in the1990s in the Myjava Hilly Land. The study was a part of more broadly aimed study of water erosion–accumulation processes operating in the post-collectivization landscape. This investigation revealed that a modern increase in muddy flood frequency is not associated with a rise in the frequency of extreme events. The increase in the intensity of water erosion in the large-scale land-use conditions resulted in an enormous increase in sediment

carried by runoff of muddy floods. This is why originally 'clean' floods, transporting negligible amounts of sediment, were transformed into muddy floods. The geomorphic effect (or in other words the thickness of the muddy layer) of individual muddy floods ranged during the 4 years of the study period from a few to 50 cm. The maximum thickness of the sediment accumulated during the whole post-collectivization period was identified in spatially limited bottoms of narrow dry valleys or in cuts incised along their thalwegs. The most interesting locality was a dry valley bottom at the hamlet of Luskovica near the village of Krajné, where it was possible to distinguish in the 105-cm thick sediment body, deposited after 1961, nine layers (Stankoviansky *et al.*, 2000), corresponding to the effect of nine muddy floods (Stankoviansky, 2002).

Measurements of deposition rates of material which has moved from slopes as far as to channels of local streams were conducted by Lehotský (1999b) in the eastern part of the Myjava Hilly Land using the dendrogeomorphological method permitted the estimation of the thickness of the deposits based on the burying of the lowest part of the tree trunks by sedimentation. The mean yearly sedimentation rate in the upper reaches was 1.3 mm, in the middle parts 2.8 mm and in the lower reaches 0.5 mm. The higher yearly values were measured in the case of a colluvial fan, namely 2.9 mm, and in a gully cut along the thalweg of dry valley, namely 4.4 mm. Although these data are semiquantitative as it is not possible to measure the sediment-contributing area, they clearly indicate that the soil redistribution is fairly active in submountainous areas.

Attention was devoted also to sedimentation in small irrigation reservoirs located on local streams, where the eroded material was transported several kilometres. Janský (1992) and Fulajtár and Janský (2001) assessed the rate of siltation in 27 reservoirs by means of a volumetric method and regression analysis (Table 1.11.2, Figure 1.11.5). The reservoirs are situated at altitudes between 135 and 380 m, half of them lying in the mountainous part of Slovakia. Their storage capacity ranges from 17 000 to 288 000 m^3, maximum water surface area from 1 to 20 ha and average depth from 0.69 to 2.54 m. The catchment area of individual reservoirs ranges from 0.8 to 28 km^2, the total area of all studied catchments is 295 km^2 and the mean area is 11.2 km^2. The calculations showed that the amount of sediments represented 4.8–83.6% of the total storage capacity of the reservoirs. The annual deposits ranged from 188 to 7 554 m^3 (with a weighted average of 2 897 m^3), giving an annual reduction of their storage volume of 0.32–9.30%. For the majority of reservoirs it was estimated that the sedimentation would fill them much sooner than is anticipated or envisaged in the period of use (100 years). Annual sediment yield, calculated by means of measured sediment volume, period of sedimentation and the catchment area, ranges in individual reservoirs between 10.4 and 442.2 $m^3\ km^{-2}$. Assuming a bulk density of 1.2 g cm^{-3}, the mean erosion rate in the studied catchments would be 34.8 t ha^{-1} with minimum and maximum rates of 2.3 and 90.6 t ha^{-1}, respectively. These values are considerably higher than the erosion rates measured by all of the above-mentioned methods. Although the siltation is mostly affected by the proportion and distribution of the forested and nonforested areas in the catchments and probably also by the effect of the total area of the catchment, the relatively high figures reflect also the role of erosion in river channels. However, not much is known about the stability of the streams and rivers in the catchments studied, which is why the proportion of riverbed material in the overall sediment volume cannot be estimated.

Systematic measurements of suspended load in selected larger rivers (Váh, Hron, Kysuca, Nitra, Hornád, Bodrog, Uh, Laborec) were carried out in 1955–72, but later only at intervals of 5–10 years. Suspended load plays a decisive role in reservoir silting. The annual sedimentation of suspended load in Slovak reservoirs is 8–10 times higher than sedimentation of the bed load (Holubová, 1997). Reservoir sedimentation in Slovakia has been discussed in numerous publications, e.g. the summary of Holubová (1997) including an exhaustive bibliography. The problem of intensive reservoir siltation is especially characteristic of the middle and upper reaches of the Váh River with numerous reservoirs, well known as the Váh Cascade. The first reservoir of the Váh Cascade at Dolné Kočkovce was built in 1935, the majority of them in the late 1940s, 1950s and 1960s and the last (at Liptovská Mara) in 1978. Reservoir siltation is related to the high rate of erosion in parts of the Váh River catchment built of rocks of medium to low resistance, namely flysch rocks. Owing to siltation, the

TABLE 1.11.2 Siltation rates in small reservoirs (Janský, 1992; Fulajtár and Janský, 2001)

Reservoir	Sub-basin	Reservoir watershed area (km^2)	Reservoir flooded area (ha)	Reservoir capacity ($10^3\ m^3$)	Nonforested watershed area (km^2)	Average annual sediment accumulation (m^3)
1. Pl. Vozokany	Hron	20.1	17	164	18.09	7554
2. Vel'ký Ďúr	Hron	10.2	10	130	10.20	3762
3. Drženice	Hron	17.5	7	98	12.25	3676
4. Mankovce	Nitra	18.0	3	50	9.00	188
5. Kolíňany	Nitra	17.0	13	106	15.30	1474
6. Čápor	Nitra	13.1	8	128	13.10	556
7. Jelenec	Nitra	11.1	7	174	5.55	1861
8. Bajtava	Hron	5.5	7	48	4.95	721
9. Dedinka	Hron	16.4	15	246	14.76	4326
10. Dubník	Hron	12.5	14	240	12.50	2360
11. Maňa	Nitra	6.2	8	169	6.20	960
12. Trávnica II.	Nitra	25.3	20	288	20.24	7478
13. Svodín	Hron	9.8	14	221	9.80	4171
14. Brezolupy	Nitra	24.0	7	90	9.60	3143
15. Nedašovce	Nitra	28.0	6	60	14.00	1174
16. Rátka	Ipel'	0.8	1	17	0.48	250
17. Bolešov	Váh	11.1	2	26	4.44	544
18. Glabušovce	Ipel'	8.7	14	180	6.96	580
19. Karná	Bodrog	2.0	2	17	1.20	578
20. Košic. Olšany	Hornád	3.5	2	25	2.80	505
21. Pol'ov	Hornád	5.1	5	75	5.10	971
22. Trstená pri H	Hornád	8.4	2	34	5.88	864
23. V. Kamenica	Bodrog	11.4	2	32	7.98	2972
24. Gem. Teplica	Slaná	3.7	14	257	2.22	1067
25. Hrušov I.	Slaná	2.6	4	36	1.82	1150
26. Nižný Žipov	Bodrog	3.5	9	146	3.50	1270
27. Bor-Továrne	Bodrog	7.5	8	203	3.75	1464

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Krpel'any reservoir lost 58%, Hričov 25% and Nosice 22% of their original volume. The total amount of sediment accumulated in the above reservoirs during the period from their construction until 1992 represents more than $12.7 \times 10^6\ m^3$, which means on average an approximately 35% reduction of their original volume (Holubová and Lukáč, 1997).

1.11.6 SOIL CONSERVATION AND POLICIES TO COMBAT EROSION

The current Slovak National Standard concerning the protection of agricultural land against both water and wind erosion contains four main groups of measures, namely organizational, agrotechnical, biological and technical, paying special attention to the technical ones. However, in the current unfavourable economic situation, technical measures are rather costly. The cheapest, and at the same time the most effective are unequivocally agrotechnical measures (Jambor, 1998).

The Soil Science and Conservation Research Institute, Bratislava, started field experiments including testing water erosion control measures in seven pilot areas in the 1990s. After this research, the Ministry of

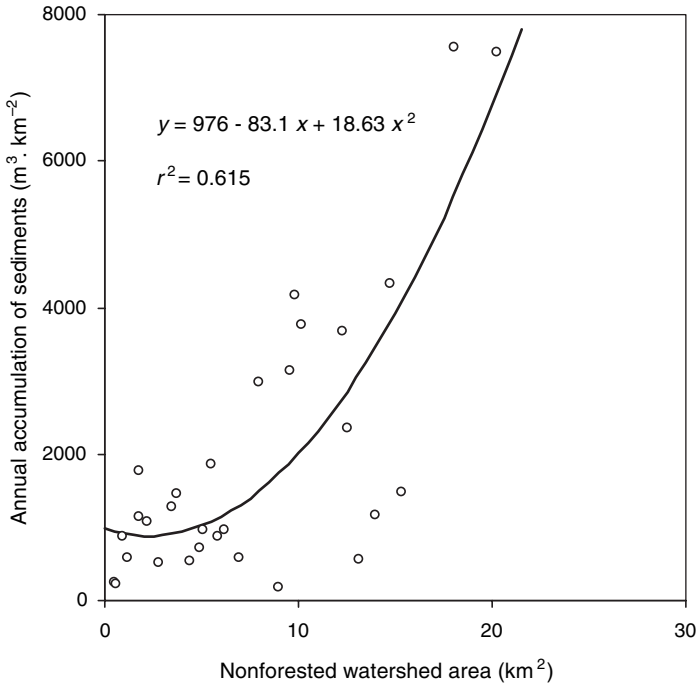

Figure 1.11.5 The relation between the rate of small reservoir siltation and non-forested watershed area (according to Janský, 1992; Fulajtár and Janský, 2001). (Reproduced by permission of the Soil Science and Conservation Research Institute, Bratislava)

Agriculture of the Slovak Republic introduced machinery necessary for conservation tillage (e.g. several tens of no-till seeding machines) and large-scale conservation tillage has started. Attention was paid especially to fields with wide-row crops that in erodible conditions were associated with the highest erosion risk. This type of conservation tillage has been applied at the national scale to approximately 140 000 ha of corn, sunflower, etc.

On the basis of the above investigations, it is possible to state that the most effective water erosion control measures in conditions of Slovakia are as follows:

- Subsoiling (0.4 m depth) in loessic soils;
- contour tillage at sites with slopes of less than 9°;
- mulching with applications of catch crop (mustard) and direct drilling;
- conservation crop rotation, where crops with a longer conservation effect (perennial crops, winter crops) are preferred;
- growing row crops and spring crops on erodible soils is admissible only in combination with conservation tillage technologies.

The most important organizational soil conservation measures recommended for conditions of Slovakia are appropriate crop rotations (Jambor and Ilavská, 1998), orientation of fields along contours, optimal shape (length 400–1000 m, width 200–300 m) and size (10–30 ha) of fields.

The application of the above soil conservation measures is most important in areas with soils possessing the highest productivity potential, namely Haplic Chernozems, Haplic Luvisols and Albic Luvisols.

Relatively extensive soil conservation measures were implemented recently in vineyards and orchards. The areal extent of vineyards increased from 17 000 ha in 1960 to 32 000 ha in 1990. A considerable part of the new vineyards was established on new, large terraces, built especially for this purpose (totally 8600 ha). Construction of these terraces represented an important contribution to water erosion control in sloping conditions.

1.11.7 CONCLUSIONS

Natural conditions of the territory of Slovakia influence its marked susceptibility to soil erosion processes, especially water erosion. This relatively high potential threat has changed, as a consequence of the historical transformation of the woodland into farmland, to the frequent to regular occurrence of actual erosion processes. However, although the conditions favouring water erosion were human interventions, periods with the most intense manifestations of this erosion process have occurred in times when the interference of humans overlapped temporally with climatic fluctuations, typically an increased frequency of extreme rainfall events. Four such periods of increased erosion in the Slovak territory in the past were identified, with the last one being combined with the Little Ice Age. Evidence of erosion processes from the period of the Little Ice Age is the network of relic, permanent gullies, often reaching a density of 2–3 km km^{-2}, locally even more, and the maximum values approach 11 km km^{-2}. The spatial distribution of gullies shows a clear linkage to the old, pre-collectivization land-use pattern. The last human interference in the form of merging of the original small private plots into large cooperateve fields as a result of collectivization in agriculture took place at the beginnings of the second half of the 20th century. These large-scale land-use changes resulted in a marked intensification of soil erosion processes. The post-collectivization period represents the fifth and continuing period of increased erosion, but the first to be conditioned exclusively by human impact.

Current water erosion occurs mainly in the agricultural land in submountainous areas and intramountainous basins, where agriculture occupies sloping land. Approximately 57% of farmland is affected and/or threatened by water erosion. Water erosion shows both on- and off-site effects. The fundamental on-site efect is the removal of topsoil and loss of organic matter, nutrients and deterioration of soil fertility. On-site effects are mostly the results of areal erosion, understood as the joint operation of sheet wash, rill and inter-rill erosion. Linear erosion, so effective in the past in connection with the formation of permanent gullies, is rather limited today. It is manifested by the formation of ephemeral gullies. It is possible to distinguish two types of ephemeral gullies: wide and shallow, cut within a cultivation layer only, and V-shaped, cut into the compacted plough pan. The most dangerous erosion events are in May and June when the combined effects of high-intensity rainfalls and poor vegetation cover on arable land occur. In some years, direct measurements of water erosion at selected sites in the agriculturally intensively utilized hilly areas showed a fluctuation of the mean soil erosion rates of around 20 t ha^{-1} yr^{-1}. However, the thick young to fresh depositional bodies observed often in the field (e.g. in the Myjava Hilly Land and in other areas) indicate that in submountainous areas erosion rates can be much higher, locally maybe markedly, and in some hot spot areas they can reach several tens of tons per hectare. As a confirmation of such a supposition, the erosion effect of a catastrophic storm event, recorded in the early 1950s close to the village of Čekovce in the Krupinská planina Mountains, can serve, with the estimation of an approximate erosion rate of 560 t ha^{-1}.

Off-site effects of water erosion are manifested by transport of eroded material to various distances and its consequent sedimentation. The major off-site effects are the pollution of water resources and siltation of reservoirs. Measurements of sedimentation in selected small reservoirs indicate fluctuations of the erosion rates in their catchments between 2.3 and 90.6 t ha^{-1}. The sediment load is high also in major Slovak rivers. Measurements in three selected reservoirs on the Váh River, built from the late 1940s to the 1960s, showed on average an approximately 35% reduction in their original volume between their construction until 1992.

The spatial distribution and hence also the effect of wind erosion are much smaller than in the case of water erosion. It occurs in lowland areas with sandy soils and frequent moisture deficit, especially in the Záhorská nížina Lowland.

The current Slovak National Standard concerning the protection of agricultural land against both water and wind erosion contains four main groups of measures, namely organizational, agrotechnical, biological and technical. The costly technical measures were more used in earlier decades, when investment in agriculture under the communist regime was much higher than it is today in order to maintain self-sufficiency in food production. At present the agrotechnical measures are considered most appropriate for Slovak conditions owing to their relative low price and high efficiency. Since the 1990s, especially conservation tillage has been encouraged. Unfortunately, the implementation of soil conservation measures at the proper level is still hindered by many obstacles.

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1.12

Hungary

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1.12.1 INTRODUCTION

Hungary (93 000 km^2) is situated in the middle of Europe between 45°48′ N and 48°35′, 16°05′ and 22°58′ E of Greenwich, surrounded by the Alpine–Carpathian–Dinaric mountain range, and occupies the inner part of the Carpathian Basin. The highest point in the country, Kékes, is 1014 m above sea level in the Mátra Mountains. The lowest point in the country is 78 m, on the Tisza river near Szeged. The country's population was 10 198 000 in 2001 and has been continuously decreasing since 1980; ca 40% are employed, of whom 6% are in agriculture, 27% in industry and 67% in other occupations.

1.12.2 PHYSICAL GEOGRAPHY

1.12.2.1 Surface Materials

Lower Carboniferous and older formations occupy a very limited area (crystaline schists of the Eastern Alps in the western part of the country, lower palaeosoic shales, phyllites and limestones north and east of Lake Balaton, migmatic granite and crystaline schists sequence in the south, limestone, shale and sandstone in the north-east). Upper Carboniferous conglomerate (sandstone–shale) outcrops can be found north-east of Lake Balaton and in north-eastern Hungary (Tokaj Mountains) and marine sediments are also known from the north-east (Bükk Mountains). The granite of Velence Hills (between Budapest and Lake Balaton) is also

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Upper Carboniferous. Permian sandstone and conglomerates can be found to the south (Mecsek and Villány Moutains) and north of Lake Balaton.

Mesosoic rocks have primary importance in the Hungarian mountains and uplands. Most of them are limestones and dolomites. Interbedded volcanics of Middle Triasic Age are known in the north (Bükk Mountains) and north of Lake Balaton (Bakony Mountains).

Limestones, marls, various clay formations and sandstones are the most important Tertiary sediments. The Miocene was the main epoch of volcanism in northern Hungary. The main rock types are andesites and pyroclastics.

Pannonian deposits are important in the hilly regions of the country and are represented by clay–marl–sand–sandstone and also by alkaline basalt volcanic pyroclastics and lavas.

Loess and loess-like sediments are the most significant Quarternary deposits. Alluvial deposits (sand, clay, gravel) of Pleistocene and Holocene age and also wind-blown sands are typical in the lowlands of the country.

From the point of view of soil erosion, it has to be emphasized that about two-thirds of the surface area of Hungary is covered by unconsolidated sediments. Loess and loess-like sediments are dominant among them.

1.12.2.2 Climate

Hungary has three climatic influences, i.e. the continental influence of the East European plains, the effect of the oceanic climate of Western Europe and the effect of the Mediterranean climate from the South. The climate therefore has a transitional character, but it can be described as moderately continental.

The mean annual temperature is between 8 and 11 °C. Most of the country has an annual temperature of 10–11 °C; the northern and western parts of the country are colder whereas the south-eastern region is warmer. The mean July temperature varies between 18 and 23 °C and the mean January temperature between 0 and 4 °C.

The north-west wind system prevails in Transdanubia and on the Danube–Tisza interfluve. East of the Tisza the north-east wind takes over and certain parts of the northern and mid-western regions of the country are characterized by northern winds. High-velocity winds are extremely important from the point of view of wind erosion.

The mean annual precipitation varies from less then 500 to more than 900 mm. The middle section of the Tisza river is the driest and the western part of the country and the highest mountain peaks are the wettest. Maximum precipitation is in June with a second maximum in October as a consequence of the Mediterranean influence. High-intensity rainfall and drought are typical of the summer. Because of the periodicity of mean annual rainfall amounts, drought periods with 400–500 mm follow wet periods of 700–800 mm yearly precipitation. Because of the basin-like character of the central part of Hungary, flooding results from rainfall and snowmelt events in the upper watershed, and also heavy rainfall and events of long duration. Snow may fall between November and March on 15–30 days.

1.12.2.3 Land Use

Land use is influenced by the fact that more than half of its area is lowland. Most of the Great Hungarian Plain was marshy until river regulation was carried out in the second half of the 19th century. Lasting consequences for the environment resulted from the river regulation measures when ca 20 000 km^2 were made available for crop cultivation. Arable land extended steadily in the second half of the 19th century. This expansion was partly motivated by increased demands for wheat. At the same time, Hungary became the second largest (after the USA) maize producer in the world. Large-scale transformations of nature at that time were not limited to the drainage of wetlands, since in the sand dune regions along the Danube and Tisza rivers, shelterbelts were planted.

TABLE 1.12.1 Land use changes in Hungary, 1895–2001 (10^3 ha)

Year	Arable land	Gardens, orchards	Vineyards	Meadows	Pastures	Agricultural land[a]	Forests	Reed	Cultivated area	Noncultivated area
1895	5103	95	175	798	1268	7439	1191	49	8678	528
1930	5587	107	214	668	1001	7577	1095	30	8702	595
1945	5567	115	215	639	962	7498	1116	29	8643	650
1950	5518	152	230	609	865	7376	1166	29	8571	728
1965	5085	319	247	419	885	6954	1422	29	8404	900
1970	5046	318	230	407	876	6875	1471	32	8378	925
2001	4516	195	93	1061	1061	5865	1772	60	7697	1606

[a]Agricultural land is arable land, gardens, orchards, vineyards, meadows and pastures combined.

With the 1920 Peace Treaty, Hungary lost its Carpathian areas (i.e. most of the forests and pastures) and some of the most fertile loess plains which were used as arable land. Disregarding a minor expansion of arable land and orchards, the pattern of land use within the borders of the present-day Hungary did not change significantly (Table 1.12.1). In the inter-War period, the categories of land use remained stable (Frisnyák, 1985).

The decade after World War II was characterized by radical social changes with implications also for land use. Industrialization attracted village-dwellers to the industrial centres and to the rapidly expanding urban agglomerations. In rural areas, changes started with a land reform. After the establishment of firm communist rule, however, new land ownership was introduced: the collectivization concentrated most of the arable land in cooperative farms. Marginal land, unsuitable for mechanized farming, did not fit in this scheme and was often left temporarily uncultivated or finally abandoned (Berényi, 1974). This explains the land-use trends observed and the gradual decrease in arable and agricultural land during the two waves of collectivization completed by 1961 (Table 1.12.1). The percentage of arable area, however, is second to Denmark in Europe today. The concentration of stock breeding and increased fodder production made many meadows and pastures superfluous. In the 1960s, a prominent expansion of forests occurred. With the rejuvenation of vineyards and the planting of large orchards, intensive branches of farming gained in importance and this is reflected in the doubling of horticultural areas between 1950 and 1965. The major changes were followed by another period of stabilization (Table 1.12.2). The declining trend of arable and agricultural land, however, continued.

In addition to the abandonment of land of marginal importance for farming, the expansion of built-up areas also contributed to the growth of nonagricultural land areas. Between 1950 and 1960, the construction of large industrial complexes, and since the 1970s motorway construction, have consumed much land. Initially, this was around the capital (Berényi, 1985) and the land used was not always of low quality. In 1961, a Land Protection Act was passed in Parliament to prevent the further loss of fertile land, a major natural asset of Hungary.

The present land-use structure is shown in Tables 1.12.1 and 1.12.2. About 77% of the Great Plain is agricultural land, of which 73% is arable. Slopes of highlands and hills gave rise to the development of traditional vine-growing and vine-producing districts. In the valleys and on the hill ridges, owing to their

TABLE 1.12.2 Changes of agricultural land use in Hungary (% of total area)

Land use	1938	1960	1985	1993	2001
Arable land	60.4	57.1	50.4	50.7	48.5
Gardens, orchards	1.3	2.0	4.8	1.4	2.1
Vineyards	2.2	2.2	1.7	1.4	1.0
Meadows, pastures	17.3	15.4	13.6	12.4	11.4
Total agricultural land	81.2	76.7	70.5	65.9	63.0

cooler and wetter climate, animal husbandry dominates, based on grazing and fodder production, and is closely associated with forestry.

The percentage of forests gradually diminished until the 1950s and was as low as 12% after World War II. The country is situated in the forest–steppe zone and the largest parts of the hilly regions were forested in the past. A considerable part of these hilly lands were deforested in 18th–20th centuries and converted into arable land. Under these circumstances, the increasing erosion hazards resulted in serious soil losses on the slopes with sedimentation and waterlogging problems in low-lying areas. As a result of reforestation, the proportion of forest is 19.2% today. Grasslands were damaged by overgrazing, by natural degradation and because of poor management. In Hungary, grasslands occur in most cases on floodplains and on peatlands with a high risk of flooding and waterlogging. They are also common in sandy and salt-affected regions with low fertility and low biomass production capacity (Stefanovits and Várallyay, 1992).

1.12.2.4 Soils

Luvisols and Cambisols are characteristic of the mountains and hills. Chernozems cover the drier and warmer lowlands, including calcarerous Chernozems, Chernozem brown forest soils, terrace Chernozems and Chernozem-type sandy soils. Phaeozems are widespread on lowlands also. Gleysols, Vertisols and Fluvisols are common in the lowest areas, i.e. on valley bottoms and on the alluvial plains. Histosols are widespread in the wetlands. The development of Solonetzes and Solonchaks is connected with the high salt content of the near-surface groundwater and of near-surface sediments. Rendzinas can be found in Transdanubian Mountains and in the Northern Uplands. Regosols and Arenosols have cover a relatively large area in sandy regions of the Great Hungarian Plain.

1.12.3 SOIL EROSION

In Hungary, as in many countries, soil is one of the most important natural resources and soil erosion studies are therefore of great importance (Stefanovits, 1977; Várallyay 1986).

Soil erosion can be considered to be one of the most significant land degradation processes in agricultural areas. Other land degradation processes, such as acidification and salinization/alkalization, compaction, destruction of soil structure, surface sealing and other chemical, physical and biological degradation processes (Várallyay and Leszták, 1990; Kertész 2001) are also important, but are not as extensive as soil erosion.

More than one-third of agricultural land (2.3×10^6 ha) is affected by water erosion (13.2% slightly, 13.6% moderately and 8.5% severely eroded) and 1.5×10^6 ha by wind erosion (Stefanovits and Várallyay, 1992) (Table 1.12.3). Moderate and strong water and wind erosion affect more than 1.7×10^6 ha (Figure 1.12.1 and 1.12.2).

TABLE 1.12.3 Soil erosion by water in Hungary

Land	Area (10^3 ha)	% of total area	% of agricultural land	% of eroded land
Whole country	9303	100	—	—
Agricultural land	6484	69.7	100	—
Arable land	4713	50.7	73.0	—
Total eroded land	2297	24.7	35.3	100
Strongly	554	6.0	8.5	24.1
Moderately	885	9.5	13.6	38.5
Weakly	852	9.2	13.2	37.4

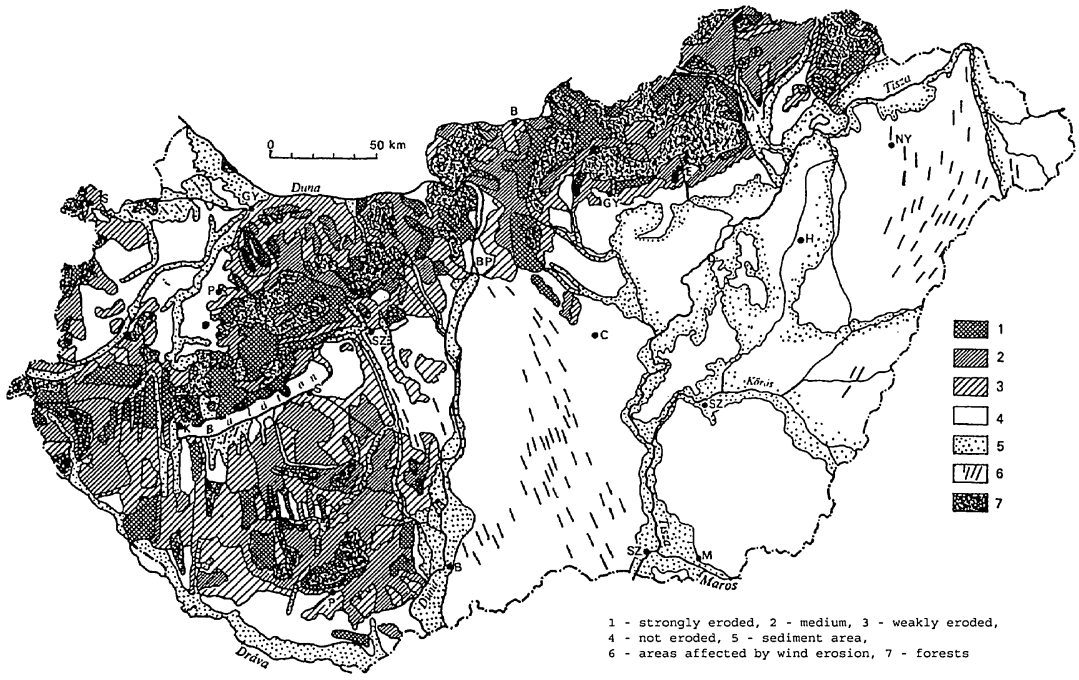
0 50 km
Duna
Tisza
Körös
Maros
Dráva
1 - strongly eroded, 2 - medium, 3 - weakly eroded,
4 - not eroded, 5 - sediment area,
6 - areas affected by wind erosion, 7 - forests

Figure 1.12.1 Soil erosion in Hungary (Stefanovits and Várallyay, 1992)

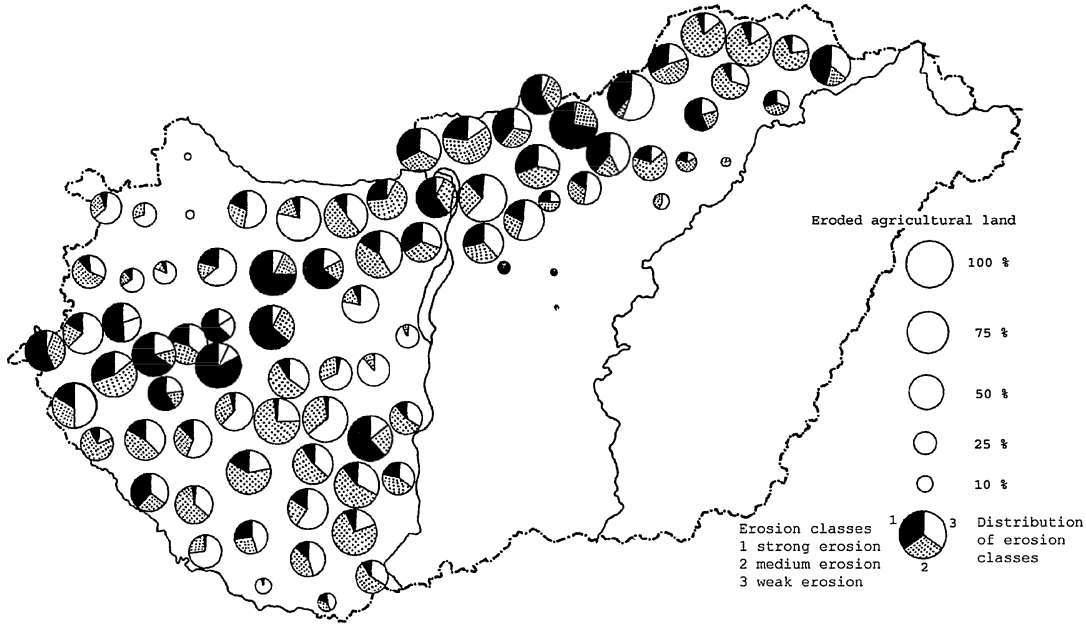

Figure 1.12.2 Distribution of eroded agricultural land in the hilly adminstrative regions of Hungary (after Stefanovits and Várallyay, 1992)

Recognizing the significance of soil erosion, a map was released in 1964 covering, however, only improved farmland (excluding nonagricultural uses, e.g. forests, urban and industrial areas, roads) and erosion by water (Stefanovits and Duck, 1964). As a result of the mapping (at a scale of 1:75 000), it was shown that 25% of the total area of the country (2.3×10^6 ha, see above) was affected by soil erosion processes. The mapping was based on the analysis of soil profiles. Soil profiles not affected by soil erosion were used as a basis for comparison in characterizing the profiles of the neighbouring area. Three stages of erosion were defined: the soil is slightly eroded if 70% of the basic profile can be detected, medium eroded in the case of 30–70% and strongly eroded if less than 30% of the basic profile remains. The compared soils are supposed to have the same bedrock and the same particle-size distribution as the basic profile. Areas effected by wind erosion are also represented on the map.

The extent of soil erosion has been estimated by many workers. According to Erődi *et al*. (1965), the rate of erosion is about $50 \times 10^6\ m^3\ yr^{-1}$ whereas some soil scientists estimate erosion at $90–100 \times 10^6\ m^3\ yr^{-1}$.

1.12.4 SOIL EROSION RESEARCH IN HUNGARY

Soil erosion assessments were mainly restricted to small areas, hillslopes or small catchments (Erődi *et al*., 1965; Csepinszky and Jakab, 1999; Dezsény, 1982, 1984; Kertész, 1987, 1993; Kertész and Góczán, 1988; Kertész and Richter, 1990, 1997; Kertész *et al*., 1993, 1995, 2000, 2001, 2002; Kerényi, 1984, 1985, 1986, 1991, 1994; Krisztián, 1992, 1998; Lóki, and Szabó 1997; Marosi and Juhász, 1992; Mattyasovszky, 1953, 1956; Máté, 1974, 1995). A short review of research activities will be provided below according to the various institutes (see also Table 1.12.4).

The first soil erosion research projects of the Geographical Research Institute (FKI) of the Hungarian Academy of Sciences started in the 1970s with plot measurements in Szomód and in Bakonynána, Western Hungary (Góczán *et al*., 1973). Plot measurements at the same site and at Pilismarót continued in the 1980s. The main objectives included runoff and soil measurements, investigations on redeposition on the slope, the role of the factors affecting erosion (slope gradient and aspect, soil and rock type, precipitation, land use and cultivation practices) and also the environmental impact of nutrients, fertilizers and pecticides (Kertész, 1987; Kertész and Góczán, 1988; Kertész and Richter, 1990).

Soil erosion projects of the Institute dealing with environmental problems of the Lake Balaton catchment date back to the late 1980s. The area selected for closer study, the Örvényes Séd stream, is a northern sub-catchment of Lake Balaton. Soil erosion studies were performed here for four years in cooperation with the German Research Foundation (Kertész, 1993; Kertész *et al*., 1993, 1995; Kertész and Richter, 1997). The main objective was the estimation of soil loss at the watershed scale using the Universal Soil Loss Equation (USLE). Research activities were extended to the southern sub-catchment. Field measurements and modelling were performed in the Tetves catchment (100 km^2) applying the USLE, EPIC and MEDRUSH models (Kertész *et al*., 2001, 2002).

Soil erosion modelling studies (FKI) were performed in the catchment of Lake Velence (604 km^2). The USLE (Wischmeier and Smith, 1978) was applied here to estimate soil loss in the catchment. The area was divided into grid cells of 30×30 m and the dominant value of each USLE factor was determined for each grid cell and a map of the factors was created.

The Department of Soil Science and Agricultural Chemistry, Szent István University, Gödöllő, has been working on soil erosion problems for several decades. A soil loss prediction map was prepared by the department (1:50 000) with the USLE model covering the central part of the northern catchment of Lake Balaton. The Department of Landscape Ecology carried out rainfall simulation experiments on seven soil types on the Balaton watershed in cooperation with Veszprém University, Department of Soil Science and Water Management, Keszthely, to determine the K factor of the USLE (Centeri, 2002). The two departments prepared a leaflet informing farmers about the behaviour of different soil types during rainfall events.

TABLE 1.12.4 Plot and watershed-scale soil erosion experiments

Site[a]	No. and size of plots	Slope gradient (%)	Genetic soil type	Crop
Kisnána Erosion Research Station (ERTI)	6 × 20 m	—	Cambisol	Forest, thicket and pasture
Pilismarót (FKI)	6, various sizes	16	Cambisol	Black fallow
Bakonynána (FKI)	6, various sizes	9	Cambisol	Black fallow
Szárítópuszta research station (SzIU-Gödöllő)	10 × 2 × 20 m	8.7	Cambisol	Winter wheat, maize and alfalfa
Siklós–Villányi hegyvidék (SzIU-Gödöllő)	10 × 160 m	—	—	Vines
Budaörsi-kamaraerdői kísérleti terület (SzIU-Gödöllő)	2 × 15 m	18	Cambisol	Vines
Cserszegtomaji kísérleti telep (Veszprém University, Keszthely)	46 m long	15	Cambisol	Vines
Kompolti (Alberta-majori research area (SzIU-Gödöllő)	12 × 5 × 60 m	15	Cambisol	Alfalfa, red clover, peas, winter barley, meadow
Szomódi runoff measuring plot (FKI)	1 × 200 m	12–15	—	Pasture, alfalfa
Péli kísérleti vízgyűjtő Udvari telepe (FKI)	1 × 5, 1 × 10, 1 × 25 m	—	—	Arable land
Balaton watershed (Veszprém University)	Varying sizes	4–10	—	Arable land, vines
Somogybabod (Veszprém University, Keszthely)	8 × 2 ha	17–19	Cambisol	Cereal, leguminous, sunflower
Balaton catchment (Somogyvár, Balatonszabadi, Tihany, Nemessándorháza) (SzIU-Gödöllő)	7 plots: 3 × 22 m	5–12	Cambisol, Regosol, calcic Chernozem, Leptosol, Luvisol	Black fallow
Somogyvár, Balatonszabadi, Tihany, Nemessándorháza (SzIU-Gödöllő)	14 plots: 2 × 6 m	5–12	Cambisol I, II, III, Regosol, calcic Chernozem, Leptosol, Luvisol	Black fallow

[a]ERTI, Scientific Institute of Forestry; SzIU, Szent István University; FKI, Geographical Research Institute.

The Department of Soil Science and Water Management, Veszprém University, has been performing very significant research using a rainfall simulator. The results were obtained within the framework of several cooperations (see, e.g., Kertész *et al*., 2002).

Veszprém University is coordinating a new erosion monitoring project on Lake Balaton watershed with three objectives: (1) clarification of phosphorus transport, (2) building up of a watershed database and (3) development of environmentally friendly farming alternatives.

Several decades ago, the Scientific Institute of Forestry, Kisnána, monitored soil loss and overland flow in six different areas, investigating among others the effect of the forest on erosion. These approaches are partly

quantitative and partly qualitative, i.e. only identifying the state of erosion of a given area. Plot and watershed scale experiments are summarized in Table 1.12.4.

1.12.4.1 Water Erosion

Owing to its relief and drainage conditions, Hungary is rather severely affected by water-erosion processes. In the mountain and hill regions, surplus runoff, the loss of soil, nutrients and fertilizers and the accumulation of washed-down material present problems. The main factors of erosion by water will be briefly reviewed.

1.12.4.1.1 Relief

The effect of relief on water erosion in Hungary is analysed according to the slope gradient categories used in Hungary. On slopes of <5%, the erosion hazard is negligible. As slopes >25% are generally forested they do not give rise to a major erosion risk (Stefanovits and Várallyay, 1992). The 17–25% slopes are either under forest or were deforested in the recent past. Most of the 5–17% slopes are used for agriculture and degraded by soil erosion to a certain extent (Krisztián, 1992).

1.12.4.1.2 Rainfall Characteristics

From the point of view of erosion, 'erosion-sensitive days' characterized by >30 mm of daily rainfall are of crucial importance (Stefanovits and Várallyay, 1992), and these occur 4–12 times per year. The percentage of precipitation falling as intense rain (>30 mm day^{-1}) during the growing season (March–October) is shown in Figure 1.12.3.

The amount of snow, the snow-cover duration and the rate of snowmelt show an extremely high spatial and temporal variability. After a cold winter when the soil is deeply frozen, quick snowmelt may result in intense surface runoff and soil erosion.

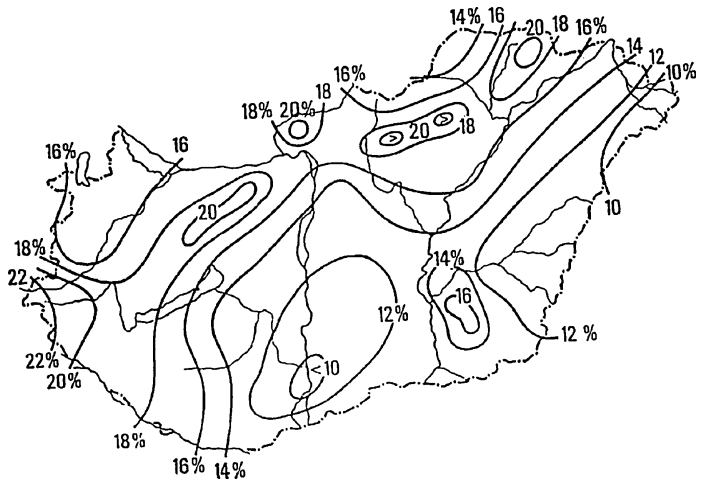

Figure 1.12.3 The percentage of precipitation falling as intense rain (>30 mm day^{-1}) during the growing season in Hungary

1.12.4.1.3 Soils

A large amount of data is available on Hungarian soils as a result of various surveys, mapping and long-term observations organized into a GIS-based soil information system (Várallyay, 1989a). Soils are generally highly erodible because soil parent material is a loose sediment, such as loess, or loess-like sediments on two-thirds of the country area. Many soil investigations have been carried out in Hungary to analyse and evaluate the influence of various soil characteristics on the rate, processes and consequences of water and wind erosion (Karácsony, 1991; Kertész and Mezősi, 1992; Kerényi, 1991; Stefanovits, 1963, 1964, 1971; Várallyay, 1986, 1989b).

1.12.4.1.4 Vegetation

There is one important thing to be mentioned about the effect of vegetation in Hungary (natural vegetation or crops). If there is ever a need for intensification of agricultural production, then the only way to do so would be via the extension of arable land. The extension of arable land can be done by deforestation of hilly regions, which would lead to increasing surface runoff, water erosion and serious loss of organic matter and plant nutrients.

1.12.4.1.5 Erosion Control

Erosion control is a principal task of farming on hillslopes. The techniques applied are agro-technological, biological and technical. The most common mechanical soil conservation practices are ridging (ridges are obliterated when ploughing on slopes of $<12\%$ and maintained on 12–17% slopes) and terracing of 17–25% slopes.

1.12.4.2 Soil Erosion

After having reviewed the factors affecting water erosion, soil erosion processes will now be dealt with briefly.

1.12.4.2.1 Sheet Erosion

Sheet erosion is an important problem on most arable land. Before the change of the regime in 1989, large arable fields were created, allowing for even greater damage by sheet erosion. Most of the wheat crop is harvested by the beginning of June, leaving extensive surfaces without vegetation cover during the most sensitive period, i.e. between July and October. Sheet erosion processes are associated with micro-solifluction and splash erosion (Kerényi, 1991). Limited infiltration due to surface compaction, soil surface sealing and crusting and the plough pan layer near the surface increase the possibility of sheet erosion on large arable fields.

1.12.4.2.2 Rill and Gully Erosion

The role of this group of processes was not properly recognized until lately and it was believed that it is mainly sheet erosion which causes damage on agricultural land. There is also historical evidence (see, e.g., Gábris *et al*., 2003) that very intensive gully erosion activity took place in the 19th century when large areas covered by loose sediments were deforested and opened for arable farming.

Special surveys were carried out to characterize gully erosion (according to the length of ravines in a given area; see Stefanovits and Várallyay, 1992):

- weakly gullied area: $<200\ \text{m km}^{-2}$ ravines;
- moderately gullied area: 200–500 m km^{-2};
- strongly gullied area: $>500\ \text{m km}^{-2}$.

1.12.4.2.3 Wind Erosion

Soil erosion by wind affects 16% of Hungary's surface. Areas of wind-blown sand occupy about 20% of the country. The thickness of the sand varies from a few centimetres to 25–30 m. Damage is primarily caused on sandy soils (e.g. on the Danube-Tisza Interfluve; see Mezősi and Szatmári, 1996) where crop yields may be reduced by up to 50%. Improperly cultivated peat soils with decomposed, powdery surfaces also have low resistance to wind erosion.

According to the estimation of Karácsony (1991), wind erosion endangers 30–40% of arable land (more than 1.5×10^6 ha). Wind erosion has been increasing during the last 2–3 decades and has degraded not only the traditionally sensitive sandy soils and peats, but also most fertile soils (Stefanovits and Várallyay, 1992).

There is a strong seasonality in deflation with peaks in early spring and summer. Improper farming practices may lead to a powdering of the soil surface or compaction, and ultimately to deflation.

In Hungary, the major factor of wind erosion is the low cohesion of a dry soil surface. The obvious preventive measure is to ensure a proper vegetation cover that reduces turbulent air motion on the surface. Rye sowing, mulching or green manuring are most often applied (Stefanovits, 1977). Inorganic materials are also suitable for sealing the soil surface (e.g. clay, bentonite injection, resins or plastic foils) (Szabó, 1977).

In order to allow mechanization after the collectivization of Hungarian agriculture, large arable fields were formed. At the present time, where wind velocity is high and droughts are frequent, small plots of 25 ha maximum separated by shelterbelts are recommended. Shelterbelts of rapidly growing trees (e.g. poplars and acacia) are preferred. Shelterbelts were introduced in Hungary following Soviet examples. They are now considered necessary on soils with poor water retention and which are liable to drought.

1.12.5 SOIL EROSION RISK MAP OF HUNGARY

The soil erosion risk assessment map of Hungary was prepared using the USLE model and ArcView GIS. Data for the factors of the USLE were taken from the following sources:

- R = rainfall erosion index (MJ mm $\text{ha}^{-1}\ \text{h}^{-1}\ \text{yr}^{-1}$), based on the rainfall erosivity map of Thyll (1992).
- K = soil erodibility factor (t ha h $\text{ha}^{-1}\ \text{MJ}^{-1}\ \text{mm}^{-1}$). Data were taken from the digital version of the Hungarian Soil Map 1:100 000 (Hungarian Academy of Sciences, Research Institute of Soil Science and Agrochemistry) and the estimation of K factors is calculated from rainfall simulation data.
- L = Slope length (dimensionless) and S = slope gradient factor (dimensionless). From the modified version (Pataki, 2000) of slope length calculation of Hickey *et al*. (1994) using the contour lines and altitude data of the 1:100 000 scale topography map of Hungary.
- C = cropping cover management factor (dimensionless). From the CORINE Land Cover 1:100 000 scale.
- P = agricultural practice factor (dimensionless). Not determined = 1.

Since sedimentation cannot be calculated by the USLE, wherever the soil properties suggested sedimentation (e.g. Fluvisols in the FAO classification or marshland with forest cover in the Hungarian classification), $K = 0$ was applied to show these soil types as potential sedimentation areas.

There are three soil loss categories on the soil erosion risk map of Hungary at the scale of 1:100 000 as shown in Figure 1.12.4:

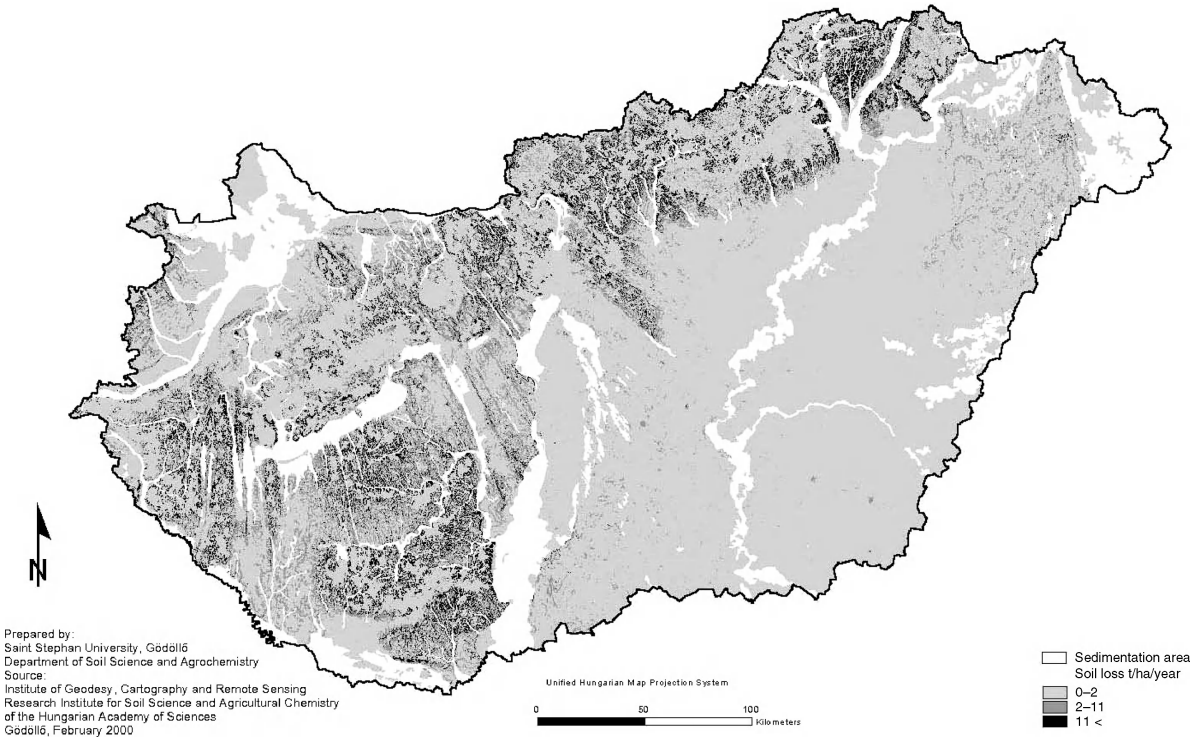

Figure 1.12.4 Predicting soil loss with the USLE in Hungary

1. 0–2 t ha^{-1} yr^{-1}. According to Hungarian estimates, this is the average rate of soil formation (Stefanovits, 1964), so if the soil loss is below 2 t ha^{-1} yr^{-1}, agricultural production can be considered sustainable.
2. 2–11 t ha^{-1} yr^{-1}. The higher value is the economically allowable nutrient loss limit for US farmers.
3. >11 t ha^{-1} yr^{-1}. Identifies areas where arable farming should not be allowed at all, or only with strict regulations.

A map was constructed first, which is not shown here, using the C factor for winter wheat on all arable land. According to this map, about 80% of the agriculturally used surface of the country is in the sustainable zone, about 14% belongs to 2–11 t ha^{-1} yr^{-1} area and 6% suffers severe erosion (>11 t ha^{-1} yr^{-1}).

The map shown in Figure 1.12.4 presents estimated soil loss on arable land with $C = 0.5$ (maize, or a crop rotation with $C = 0.5$).

1.12.6 SOIL CONSERVATION IN THE CONTEXT OF POST-WAR HUNGARIAN AGRICULTURE

According to Erődi *et al*. (1965), Krisztián (1992) and Stefanovits and Várallyay (1992), three main periods of post-War development of Hungarian agriculture can be distinguished, in addition to a fourth period which began after the change of regime in 1989.

In the period 1940–50, small-scale farming with low input levels, low yields and serious erosion damage was typical. Accelerated erosion was mainly due to downslope cultivation on small, elongated plots. The first collectivization programme of the early 1950s did not survive the 1956 revolution.

A successful collectivization programme was completed in the early 1960s. As a result, about 25% of the land was owned and used by the State farms, 65% was used by the cooperatives (and still owned – theoretically – by the members of the cooperatives) and less than 10% was owned and used privately.

A spectacular agricultural development started roughly 10 years after the collectivization programme had been completed. The communist system wanted to prove that the large-scale, collective sector is more efficient than the small-scale private sector. Economic regulations were developed and introduced accordingly (high-rate State subsidies for collective agriculture, credit and pricing policies). The efforts proved to be successful at the beginning (Stefanovits and Várallyay, 1992). Crop yields doubled or tripled because of the new, intensive varieties and the high amount of mineral fertilizer applied, accompanied by full mechanization and integrated pest management. Special attention was paid to the introduction of soil conservation practices in the hilly and mountainous regions of Hungary, including the necessary measures for soil conservation research. Demonstration farms were established using soil conservation practices, such as soil-protecting cropping patterns and crop rotation, strip cropping, contour ploughing and terracing (Stefanovits and Várallyay, 1992).

The problems of the State-subsidied system, including the lack of efficiency, economic, quality and environmental aspects, became obvious in the third period starting in the 1970s. Huge agricultural fields (100–150 ha) were established even on hillslopes, sacrificing the windbreaks, forest shelterbelts, grass strips and soil conservation practices (terraces, contour ploughing). Arable land (including large-scale corn production) was extended to hilly regions. Very serious environmental consequences started to develop, such as an increasing rate of water and wind erosion, soil acidification, salinization/alkalization, structural destruction and compaction of soils, pollution of soils, nitrate contamination of drinking water supplies and P-load of surface waters with their unfavourable consequences (Stefanovits and Várallyay, 1992). The latter led to an increasing rate of eutrophication and sedimentation of lakes so that off-site problems of soil erosion started to become more and more important. The ecosystem of Lake Balaton is severely threatened by accelerated eutrophication (Kertész and Richter, 1997).

The most recent period of Hungarian agriculture started at the end of the 1980s. On the one hand, the environmental consequences of the over-intensive, subsidied and noneconomic third period led to the collapse of the large-scale communist agriculture. On the other hand, the change of regime created totally new conditions for a new agriculture based on the market economy.

1.12.7 SOIL CONSERVATION POLICY

Soil conservation became part of State agricultural policy (Stefanovits, 1977; Várallyay and Dezsény, 1979). In 1957, the National Soil Conservation Council (later renamed Amelioration Council) was set up to coordinate conservation activities and to identify priorities in this field.

Soil conservation planning began at three levels:

1. Plans for the whole country or for selected regions or watersheds were made at 50 000 to 100 000 scales.
2. Smaller regions and partial watersheds were surveyed and recommendations were made at 1:50 000 to 1:25 000 scales.
3. For individual farms, scales of 1:25 000 to 1:10 000 were preferred.

The protection of agricultural land had been covered in the 1961/VI Act. It was the partly bureaucratic constraints on land ownership and partly the wastefulness of land use in terms of the conversion of prime land

to other purposes and the resulting steady reduction of cropland that necessitated comprehensive legislation on land.

The 1987/I Act (popularly called the Land Codex) includes provisions on the soil-conserving cultivation of land, according to the physical endowments and actual land use.

Since 1989, an important re-privatization process began and the role of State subsidies became less and less important. Chapter VI of Law 55 (1994) regulates soil conservation. It determines the major threats that soil must be protected against, water and wind erosion, extreme moisture conditions, salinization/sodification, acidification and other physical, chemical and biological degradation processes. According to this law, soil conservation is a joint task of the State and the land user. The government has to maintain a database and monitoring system on soil quality, to develop legislative and economic measures, to create and maintain a soil conservation authority, to prepare and to implement a national strategy of soil protection and to promote and fund research and development programmes. As far as the land user is concerned, the law prescribes soil conservation practices to be applied by farmers, but lower level legislation containing detailed measures is still missing or under preparation. Under the changing property and land use situation, the 1994 law was urgently needed to be adapted to market conditions. Agricultural land had to become the guarantee for mortgage credit. On the other hand, the quality of agricultural land had to be protected and the legal basis for this had to be described and provided for.

Apart from the 1994 law on agricultural land, some other laws should be mentioned that are in close relationship with agricultural land and soil, i.e. Law 57 (1995) on water management, Law 53 (1995) on general rules of environmental protection, Law 21 (1996) on regional development and Law 56 (1996) on forests and their protection. Law 56 (1996) must be applied in accordance with the provisional rules on nature protection, agricultural land/soil protection, plant protection and hunting practices. The soil under the forest is considered also to be part of the forest.

Governmental Decree No. 49/2001 (IV.3) on the protection of waters against nitrates from agricultural sources describes the rules of good agricultural practice including some measures on erosion control. Application of these measures is obligatory in nitrate vulnerable zones to be checked by the Soil Conservation Service. Farmers may apply for subsidies for deep loosening according to Ministerial Decree No. 3/2003 (I.24) of the Ministry of Agriculture and Rural Development.

Last, but not least, the Hungarian Agri-environmental Programme provides financial help for farmers who are willing to apply environmentally friendly farming practices. The programme was launched in 2002. It contains a sub-programme on protection against erosion, planned to be started in 2004.

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1.13

Romania

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1.13.1 INTRODUCTION

Romania covers 237 500 km^2 of south-eastern Europe and consists of three major relief units: the Carpathian Mountains and Sub-Carpathians (36%), the hills and plateaus (34%) and the plains (30%). Within its boundaries live 22.3 million people.

Hot summers and cold winters, variability in the distribution of rainfall and fluctuating length of the growing season typify the Romanian continental–temperate climate. However, there is a clear transition between the central European climate in the centre and west and the east European climate in south and east of the country.

Mean temperatures decrease with increasing elevation, from 10 °C on plains and 8–9 °C on plateaus to 7–8 °C on lower mountains of 700–1200 m elevation and –2.6 °C on high mountains with elevations around 2500 m. The minimum temperature of −38.5 °C was recorded on 25 January 1942 near Brasov, in the central part of Romania and the maximum temperature of +44.5 °C on 10 August 1951 near Braila, in the southeast region. Average annual precipitation varies from about 360 mm at lower elevations in the Danube delta to 1450 mm in the high mountains.

In hilly areas, as a result of erosion, mostly clayey, sandy Tertiary layers outcrop. Mollisols (Chernozems, grey forest soils) and Argiluvisols (reddish-brown soils, brown soils, brown–luvic and Luvisols) are among the most common soils and have been used for crop production.

According to an inventory undertaken in 1980 by the Institute of Geodesy, Photogrammetry, Mapping and Land Organization, agricultural land in Romania averaged about 63% of the total (Table 1.13.1).

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TABLE 1.13.1 Land use in Romania

Land use	Surface	
	10^6 ha	%
Arable	9.833	41.4
Pasture (grazing land)	4.467	18.8
Vineyard	0.306	1.3
Orchard	0.357	1.5
Agricultural total	**14.963**	**63.0**
Woodland	6.568	27.7
Waters and marshes	0.796	3.3
Roads and railroads	0.375	1.6
Yards, construction areas	0.655	2.8
Unproductive (abandoned)	0.393	1.6
Nonagricultural total	**8.787**	**37.0**
Total	**23.750**	**100.0**

1.13.2 MAGNITUDE OF SOIL EROSION IN ROMANIA

Erosion surveys started in 1947 in Buzau County and reached a climax when Florea *et al*. (1977) released the map of soil erosion in Romania at 1:500 000 scale. The map in Figure 1.13.1 shows that the potential for soil erosion caused by water is far more severe than for wind erosion. Agricultural land subjected to water erosion averages 45.6% of the total, whereas wind erosion is a potential threat on only 1.4% located in the south. Motoc (1983) provided a similar value with a potential for water erosion on 42.6% of the total Romanian agricultural land. Of those 6.4×10^6 ha, 2.6×10^6 ha are cropland, 3.4×10^6 ha are pasture and 0.4×10^6 ha are orchard and vineyard.

Figure 1.13.2 shows erosion rates in different areas of the country (Motoc, 1983). The estimated peak erosion rate rises to 30–45 t ha^{-1} yr^{-1} and it occurs in the curvature of the Sub-Carpathians. Slightly lower erosion rates are found in the southern Sub-Carpathians, the Getic Plateau, Moldavian Plateau and the Transilvanian Plateau. The classes with moderate and high rates of erosion (9–10 t ha^{-1} yr^{-1}) are predominant. Motoc (1983) also devoted special attention to the sediment source. Tables 1.13.2 and 1.13.3 show the contributions of both land use and the erosion processes to the total erosion.

Motoc's (1983) estimates are based on various sources: the map of soil erosion in Romania (Florea *et al*., 1977); a map of suspended sediment concentration in Romania (Diaconu, 1971); a model for estimating soil erosion in small catchments where total erosion is the sum of surface erosion, gully erosion and erosion from landslides (Motoc *et al*., 1979); runoff plot data; studies of soil erosion in different catchments, such as Arges and Putna; soil conservation projects in representative catchments provided by ISPIF (Institute for Land Reclamation Studies and Designs); an inventory of the agricultural land in Romania undertaken in 1980 by IGFCOT (Institute of Geodesy, Photogrammetry, Mapping and Land Organization); an inventory of the forestry land in Romania undertaken in 1981 by ICAS; and a pedoclimatic division and land classification of the Romanian agricultural land by ICPA (Research Institute for Pedology and Agrochemistry) in 1975.

These data show the different sources contributing to the gross erosion. Of the 126.6×10^6 t, 106.6×10^6 t, which equates to 84% of the total, is delivered by agricultural land. The low vineyard and orchard input results from the setting of plantations under conservation treatments for the last 30 years. Annual sheet and rill erosion rates average 61.8×10^6 t, which is twice as great as the next highest rate (29.8×10^6 t for gully erosion). Therefore, sheet and rill erosion and gully erosion are the most important

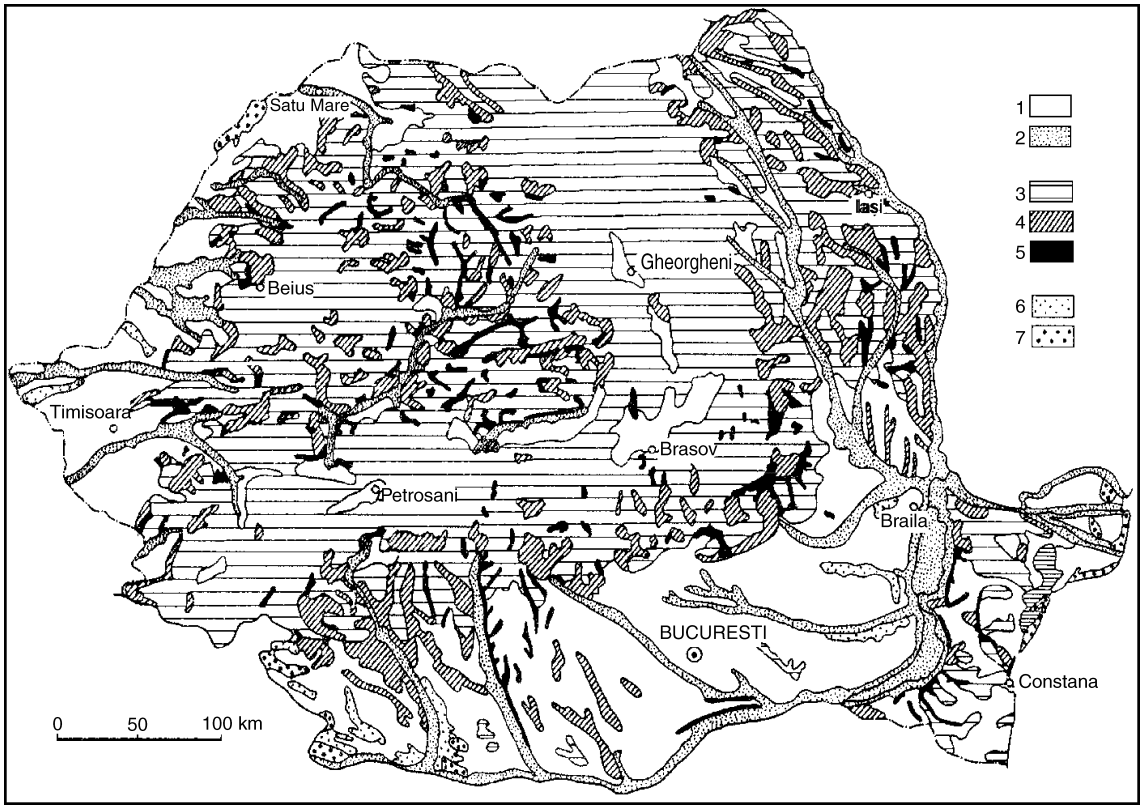

Figure 1.13.1 Soil erosion map of Romania: 1, erosion free land with no flooding risk; 2, erosion-free land at risk from flooding and siltation; *land subjected to water erosion*; 3, slightly eroded soils; 4, moderate to strongly eroded soils; 5, severe to excessively eroded soils; *land subjected to wind erosion*; 6, moderate to strongly eroded; 7, severe to excessively eroded. (Reproduced from Florea N. *et al*., 1977, with permission from N Florea)

contributors to gross erosion, whereas landslides have a lesser input. Moreover, the sediment delivery ratio averages 0.35, equating to 44.5×10^6 t yr^{-1} at the national scale (Motoc, 1984).

According to Ichim and Radoane (1987) and Ichim *et al*. (1998), the sediment yield from the 1100 km^2 Putna basin is 12.5 t ha^{-1} yr^{-1} from flysch strata and 45.4 t ha^{-1} yr^{-1} from Neogene molasses layers. More than 50% of the sediments originating in small catchments are deposited in third-order basins of the flysch area, whereas in the Sub-Carpathians only 30% is stored.

For understanding basic processes, studies of dispersed overland flow, rill-flow and the flash streamflow were undertaken. Runoff plots were set up under different conditions in some agricultural research stations at Cean-Turda (Motoc, 1975), Perieni-Barlad (Motoc *et al*., 1998; Ionita, 2000b), Podu-Iloaiei (Dumitrescu and Popa, 1979), Aldeni (Ene, 1987) and Bilcesti (Teodorescu and Badescu, 1988). Table 1.13.4 summarizes and illustrates the substantial differences in erosion rates reported for each land use. Of those stations, the Perieni-Barlad within the Moldavian Plateau is the most interesting. Data collected here over a 30-year period indicate the following (Ionita, 2000b):

- Mean annual precipitation is 504.3 mm and precipitation which causes runoff and erosion occurs during the crop-growing months of May–October.

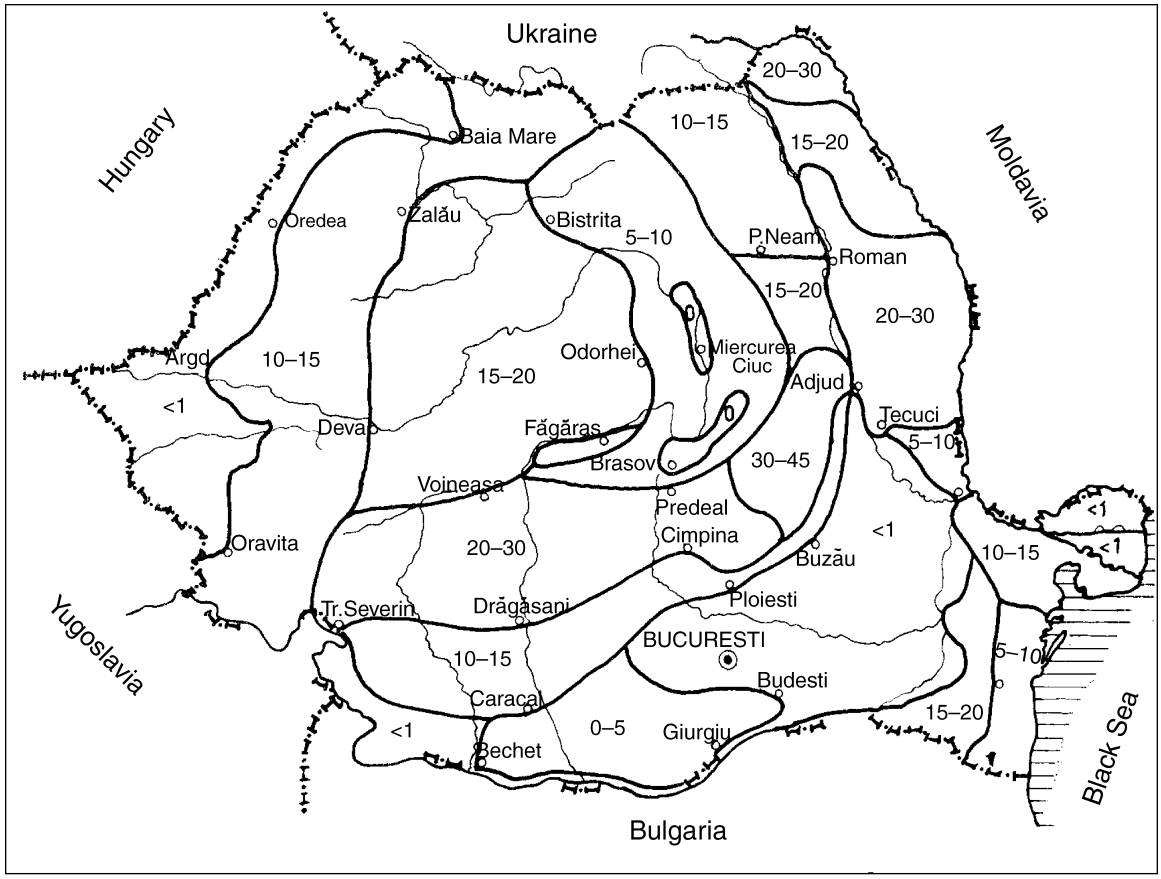

Figure 1.13.2 Total erosion on agricultural lands in Romania ($t\,ha^{-1}\,yr^{-1}$) (Reproduced from Motoc M, 1983, with permission of M Motoc)

TABLE 1.13.2 Total erosion by land use in Romania

Land use	Total erosion			
	$10^6\,t\,yr^{-1}$		%	
Arable	28.0	26.2	24.7	22.3
Pasture (grazing land)	45.0	42.2	39.6	35.7
Vineyard	1.7	1.6	1.5	1.2
Orchard	2.1	2.0	1.8	1.7
Unproductive (Abandoned land as gullies)	29.8	28.0	26.4	23.6
Agricultural land total	**106.6**	**100.0**	—	—
Woodland	6.7	—	6.0	5.3
Total	113.3	—	**100.0**	—
River bank and localities erosion	12.7	—	—	10.2
Total	**126.0**	—	—	**100.0**

TABLE 1.13.3 Total erosion by types of processes

Process	Total erosion		
	10^6 t yr^{-1}	%	
Sheet and rill erosion	61.8	54.5	49.0
Gully erosion	29.8	26.4	23.6
Landslides	15.0	13.1	11.9
Gully erosion and landslides in woodland	6.7	6.0	5.3
Total	113.3	100.0	—
River bank and localities erosion	12.7	—	10.2
Total	**126.0**	**—**	**100.0**

- About 26% (133.5 mm) of the annual precipitation induced runoff/erosion is on continuous fallow and 18.5% (93.5 mm) on maize.
- Runoff ranges from 36.5 mm under continuous fallow with a peak of 12.0 mm during July and 17.7 mm under maize with a peak of 6.5 mm during June.
- Soil loss is averaging 33.1 t ha^{-1} yr^{-1} for continuous fallow with a peak of 12.8 t ha^{-1} during July and 7.7 t ha^{-1} yr^{-1} for maize with a peak of 3.7 t ha^{-1}during June (Figure 1.13.3).

According to Motoc *et al*. (1998) and Ionita (2000b), data collected from a continuous fallow plot and processed by using a 3-year moving average revealed that over the period 1970–99 there were three peaks. The centre of those highest values is placed at 1975, around 1988 and 1999 (Figure 1.13.4).

By processing such data, Motoc (1960, 1983) developed a quantitative model to evaluate soil loss by sheet-rill erosion. It is the same type as Wischmeier's model.

The *Hi*15 indicator proposed by Stanescu *et al*. (1969), where *H* is the amount of precipitation and *i*15 the intensity of the rainstorm of 15-min duration, was of value in running this model. The *Hi*15 index for rainfall aggressiveness can to be calculated more easily than, and has a similar value to, the rainfall erosion index proposed by Wischmeier (1959) for the USA.

Concerning gully development and concentrated flows, research carried out by Ionita (1998, 1999, 2000a, 2003) in the Moldavian Plateau of eastern Romania is of particular interest. In order to obtain a clear image of the development of continuous gullies, 13 gullies were first sampled near the town of Barlad. Most have catchments smaller than 560 ha. Linear gully head advance, areal gully growth and eroded material rates were quantified for three periods (1961–70, 1971–80 and 1981–90). The results indicate that gully erosion has decreased since 1960 (Figure 1.13.5). This decline in gullying results from changes in rainfall distribution and the increased influence of soil erosion control. The mean gully head advance of 12.5 m yr^{-1} between 1961 and 1990 was accompanied by a mean areal gully growth of 366.8 m^2 yr^{-1} and a mean erosion rate of 4168 t yr^{-1}. Most of these catchments exhibit average values of soil loss due to gullying ranging from 10 to 40 t ha^{-1} yr^{-1}.

The average annual regime of gullying was documented through a periodic survey of six continuous gullies over the period 1981–96 and showed a pulsatory development. It exhibited great fluctuations that ranged from stagnation to average annual peak values of 19.1 m gully head advance and 304.0 m^2 areal gully growth during 1988. The four rainy years of 1981, 1988, 1991 and 1996 contributed 66% of the total gully growth. Another main finding of this 16-year stationary monitoring was that 57% of the total gullying occurred during the cold season, with the remainder during the warm season (Figure 1.13.6). The critical period for gullying covers the 4 months between 15–20 March and 15–20 July.

TABLE 1.13.4 Runoff plot data in Romania

Site	Years	Land use	Slope (%)	Length (m)	Area (m^2)	Mean annual precipitation (mm)	Mean annual runoff (mm)	Mean annual soil loss ($t\ ha^{-1}\ yr^{-1}$)	Soil type	Latitude (° N)	Longitude (° E)
Perieni,	1970–99	Winter wheat	12	25	100	504.3	5.2	0.70	Mollisol,	46.16	27.37
Vaslui		Maize					17.7	7.74	loamy		
County		Continuous fallow					36.5	33.10			
Podu Iloaiei,	1965–77	Winter wheat	16	25	100	532.9	6.4	2.70	Mollisol,	47.12	27.16
Iasi County		Maize					20.5	17.20	loamy–		
		Sunflower					21.0	20.80	clayey		
Aldeni,	1975–81	Winter wheat	18	40	200	389.6[a]	31.1	7.00	Mollisol	45.19	26.46
Buzau		Maize					66.2	26.60			
County		Fallow					93.5	44.80			
Bilcesti,	1968–85	Apple trees,	25	20	120		36.9	13.80	Brown	45.17	25.06
Arges		no terraces		40	240		39.7	27.30	luvic,		
County		Apple trees	25	20	120	737.5	27.0	5.10	loamy–		
		under terraces		40	240		14.0	2.80	sandy		
Cean-Turda,	1950–59	Winter wheat	7	25	100		—[b]	0.67	Mollisol,	46.40	24.00
Cluj County		Maize						7.90	loamy–		
		Winter wheat	12	25	100	542.3	—[b]	0.90	clayey		
		Maize						12.40			

[a]Over 1 April–30 September.
[b]No available data.

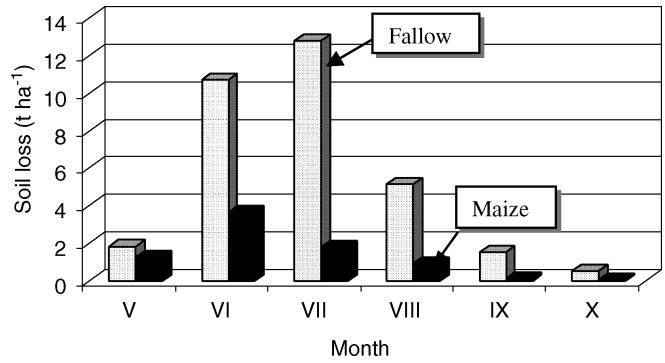

Figure 1.13.3 Mean monthly soil loss under fallow and maize plots at Perieni, Romania (1970–99)

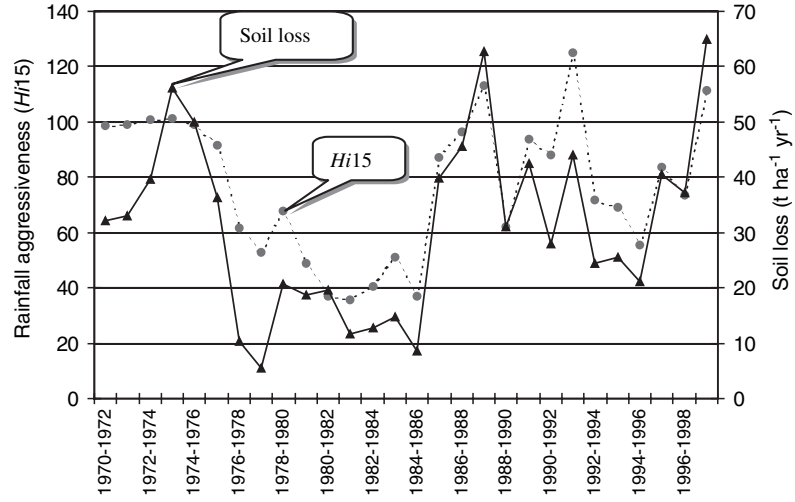

Figure 1.13.4 Rainfall aggressiveness (*Hi*15) and soil loss under continuous fallow at Perieni, Romania (1970–99)

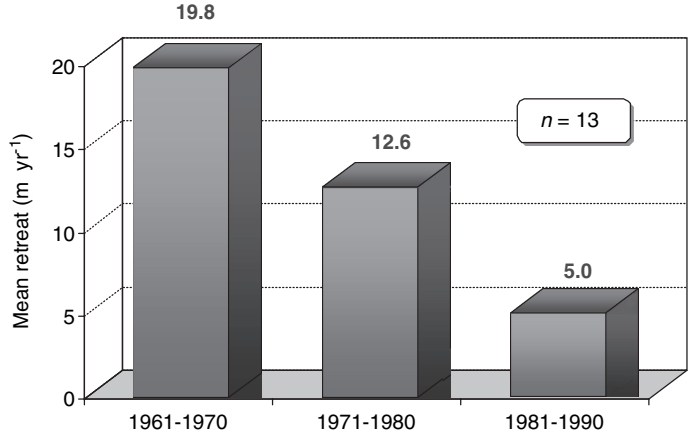

Figure 1.13.5 Measured gully head retreat in the Moldavian Plateau, Romania (1961–90)

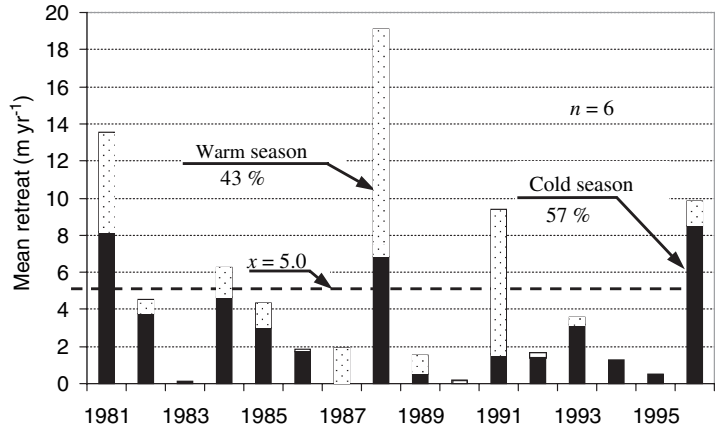

Figure 1.13.6 Measured mean rate of gully head retreat in the Moldavian Plateau, Romania, between 1981 and 1996

As regards discontinuous gullies, results have indicated that during a variable period of 6–18 years the mean gully head advance was 0.92 m yr^{-1} and ranged from 0.42 to 1.83 m yr^{-1}. The mean areal gully growth was 17.0 m^2 yr^{-1}, varying between 3.2 and 34.3 m^2 yr^{-1}. Both values indicate a slow erosion rate.

Field measurements performed in small catchments within the Moldavian Plateau during flash streamflows showed two types of sediment delivery, synchronous and asynchronous. In the synchronous case there is almost simultaneous production and removal of debris. In the asynchronous case there is a preparatory stage during late winter and early spring prior to removal of the debris. The synchronous scenario occurs rarely and is mostly associated with quick thawing, and gives very high sediment concentrations, exceeding 300.0 g l^{-1} at the basin outlet and low values, up to 40.0 g l^{-1} upstream of gully heads in the upper basin. Gullying is the major sediment source. The asynchronous scenario commonly occurs and is characterized by higher water discharges and fluctuating sediment concentration (Ionita, 1998, 1999, 2000a).

A multiple regression model was proposed by Radoane *et al*. (1995, 1999) for assessing the rate of the gully head advance between the Siret and Prut rivers:

$$R_a = aA^bL^cE^dP^e$$

for the gullies on marls and clays and

$$R_a = a + bA + cE + dL + eP$$

for the gullies on sandy layers, where R_a = the rate of the gully head advance (m yr^{-1}), A = the drainage basin area upstream of the gully head (ha), L = gully length (m), E = the relief energy of the drainage basin (m) and P = drainage basin slope (m per 100 m). By processing data from 38 mainly discontinuous gullies, the estimated rate of gully head retreat was over 1.5 m yr^{-1} on sands and less than 1 m yr^{-1} on marls and clays.

Mircea (1999, 2002) evaluated the rate of the gully head advance as ranging from 1.75 to 6.70 m yr^{-1} within some small catchments of the Buzau Sub-Carpathians over the period 1962–89 using MODPERL (MODel de Prognoza a Evolutiei Ravenelor in Lungime, a model for predicting the development of gullies in length) that has the following functional form:

$$R_{\mathrm{ar}} = [a + (bq_{10\%} + ch + di_{\mathrm{fv}})]CUC_{\mathrm{s}}$$

where R_{ar} = the annual rate of the gully head retreat (m yr^{-1}), $q_{10\%}$ = the unit discharge at 10% frequency (m^3 s^{-1} m^{-1}), h = the depth of the headcut (m), i_{fv} = the slope of the valley-bottom upstream of the gully head (%), C = the erodibility factor (0–1), U = the layer moisture factor ($\leq$1) and C_s = conservation practices factor within the basin (0–1). Model calibrating and testing resulted in underlining the strong influence of unit discharge and the low influence of both the headcut depth and the slope of the valley-bottom on the gully head advance.

The denudation rate by landslides was evaluated by Balteanu (1983) as ranging from 0.6 to 73.8 mm yr^{-1} in the Buzau Sub-Carpathians and by Pujina (1997) averaging 36.0 mm yr^{-1} within the Barlad Plateau between 1968 and 1992.

Romania is a country where the tradition of dam construction is very old. Among the 80 members of the International Committee of the Big Dams, Romania occupies the 19th place with respect to the number of 'big dams' (over 15 m height) and the ninth place in Europe. The total number of big dams is 246, and almost half are dams under 40 m height. About 90% of the existing reservoirs have storage capacities under 200×10^3 m^3, and half of them are under 20×10^6 m^3. There are some dam reservoirs that have been functional for centuries, such as those in Banat Mountain or Metaliferi Mountains, but there are also lakes that became silted in a short period of time.

Ichim and Radoane (1986) and Radoane and Radoane (2005) came to the following conclusions:

- Over an average of 15 years a volume of about 200×10^6 m^3 of sediments has been deposited in the reservoirs within the interior rivers, of which the Arges and Olt rivers contributed almost 50% of the total.
- The largest annual silting rates are associated with the lakes in the Sub-Carpathians such as on the Arges River (Pitesti 15.7%, Bascov 11.7%, Oiesti 9.5%, Cerbureni 7.3% and Curtea de Arges 5.3%) and the Siret River (Galbeni 10.6%).
- Average annual rates of faster silting have been recorded also at the first lakes, built on the Olt river (Govora 8.3%, Rm. Valcea 5.6% and Daesti 4.9%), Bistrita river (Pangarati 3.5%) and the Ialomita river (Pucioasa 2.6%).
- Low rates of silting have been assessed in the big reservoirs of Izvoru Muntelui (0.03%) and Vidraru (0.04%).
- The silting time of 50% of a reservoir's volume is reduced to less than 100 years for the lakes that lie in areas with high sediment yield (Sub-Carpathians, plateaus and piedmont). In other words, only 57 reservoirs have enough silting time to justify the investment and the significant environmental changes.

Measurement of the caesium-137 content of sediments established the rate of sedimentation in 15 reservoirs of the Moldavian Plateau (Ionita *et al*., 2000). The estimated mean values vary between 2.6 and 7.9 cm yr^{-1}- with an average value of 4.6 cm yr^{-1}after April 1986. The shape of the caesium-137 depth profile was used as the main approach. Taking into account that the standard pattern is in the form of a cantilever and based on the burial magnitude of the caesium-137 peak derived from Chernobyl, two main patterns of reservoir sedimentation were identified, shallow and deep buried cantilever.

The caesium-137 technique has also been used effectively in areas of deposition of gully sediment to provide a chronological measure of gully development (Ionita and Margineanu, 2000). The mean sedimentation rate is 4.4 cm yr^{-1} between 1963 and 1996 and 2.5 cm yr^{-1} after 1986 in the short successive discontinuous gullies. In the case of long discontinuous gullies, these values are almost double.

1.13.3 SOIL CONSERVATION

Soil erosion and associated water runoff increase short-term farm production costs per unit of harvested crop in a variety of ways. In most cases, to correct runoff problems and to bring erosion under control,

farmers have several options: use conservation practices, change land use or crop rotation and alter field boundaries.

Before 1960, the traditional agricultural system on the hills of Romania consisted of up-and-down-slope farming. Most of the land, accounting for roughly 85% of agricultural acreage, was split into small plots, each of less than 1 ha in size. Except in local areas, there was no concern about the soil erosion threat and a minimum awareness of conservation practices. After 1960, the area comprising those small plots was turned into cooperative farms. The remaining 15% of the agricultural land, which belonged to proper farmers (as regards the ownership size), was changed to State farms.

After several decades of quiescence, many new, innovative research studies on soil erosion control have been initiated (Motoc *et al*., 1975, 1992; Nistor and Ionita, 2002). For the nation as a whole, the first priority consisted of implementing one or more conservation practices. The first important objective was to plough on or nearly on the contour as one of the simplest of conservation methods. Then, based on the experience gained by the Central Research Station for Soil Erosion Control of Perieni-Barlad, some representative farms under conservation practices were set up on 65 000 ha. By the end of 1989, as much as 2.2×10^6 ha, equating to 30% of agricultural land at risk of erosion, was adequately treated with conservation measures.

The new land property law No. 18/1991 includes two provisions that do not encourage the extension of conservation measures (Motoc *et al*., 1992; Nistor and Ionita, 2002). One of these stipulates that land reallotment has to be done as a rule in the old locations. In most cases, this means that the plots will be up-and-down slope. The second refers to the successors' right up to the fourth degree. Under these circumstances, the rate of land division increased and it is higher than before World War II. Another law, No. 1/2000, was promulgated and is focusing on the forestland division for private ownership. The major effect of the earlier mentioned laws is the revival of the traditional agricultural system with up-and-down slope farming. Another problem over the last decade is that the state ceased funding soil erosion control and such an investment does not represent a priority for landowners.

The depth distribution of caesium-137 in recent sediments in the Bibiresti reservoir within the upper Racatau basin of 3912 ha produced evidence of a doubling in erosion/deposition rates after a contour farming system was converted to a traditional up-and-down slope system (Ionita and Margineanu, 2000). Therefore, it might be concluded that real soil erosion control in Romania took place over the 30-year period from 1960 to 1990.

1.13.4 CONCLUSIONS

Romania is a central and eastern European country that presents various forms created by land degradation because of its natural conditions. Agricultural land subjected to water erosion averages 43% of the total, whereas wind erosion is a potential threat on only 1.4%.

The total erosion was estimated at 126.6×10^6 t yr^{-1} and of this, 106.6×10^6 t yr^{-1}, million which equates to 84% of the total, is delivered by agricultural land. Inter-rill and rill erosion and gully erosion are the most important contributors to gross erosion since their specific erosion rates average 61.8×10^6 t yr^{-1}. The sediment delivery ratio averages 0.35, equating to 44.5×10^6 t yr^{-1} at the national scale.

Before 1960, the traditional agricultural system on the hills of Romania consisted of up-and-down slope farming. By the end of 1989, as much 2.2×10^6 ha, equating to 30% of agricultural land with erosion potential, was adequately treated with conservation practices.

The new land property law No. 18/1991 includes two provisions that do not encourage the extension of conservation measures. The major effect of this law is the revival of the traditional agricultural system, up-and-down slope farming.

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1.14

Bulgaria

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1.14.1 INTRODUCTION

Development of sustainable agricultural systems to satisfy the present and the future needs of mankind requires knowledge on constraints and the potential of land resources. The UNEP Project GLASOD (GLobal Assessment of SOil Degradation) recognized erosion by water as the most important soil degradation type, representing more than half of all soil degradation (Oldeman *et al*., 1991). Estimates by Bot *et al*. (2000) showed that 34.0% of the soil constraints in Europe are associated with erosion risk. The corresponding estimate for Bulgaria is 32% (Bot *et al*., 2000). This chapter aims at presenting an overview of the major erosion processes affecting all Bulgarian land.

1.14.2 LOCATION AND PHYSICAL GEOGRAPHY

Bulgaria is located in south-eastern Europe and occupies the eastern part of the Balkan Peninsula between 41°14′ and 44°13′ N and 22°21′ and 28°36′ E. The area of the country is 110 993.6 km^2. Bulgaria borders to the north Romania, to the south Greece and Turkey, to the west Serbia and Macedonia and to the east the Black Sea (Figure 1.14.1).

The main characteristic of Bulgaria's topography is alternating bands of high and low terrain that extend east to west across the country. From north to south, those bands are the Danubian Plateau, the Balkan Mountains (giving the name to the whole peninsula, called also Stara Planina in Bulgarian, which means 'old

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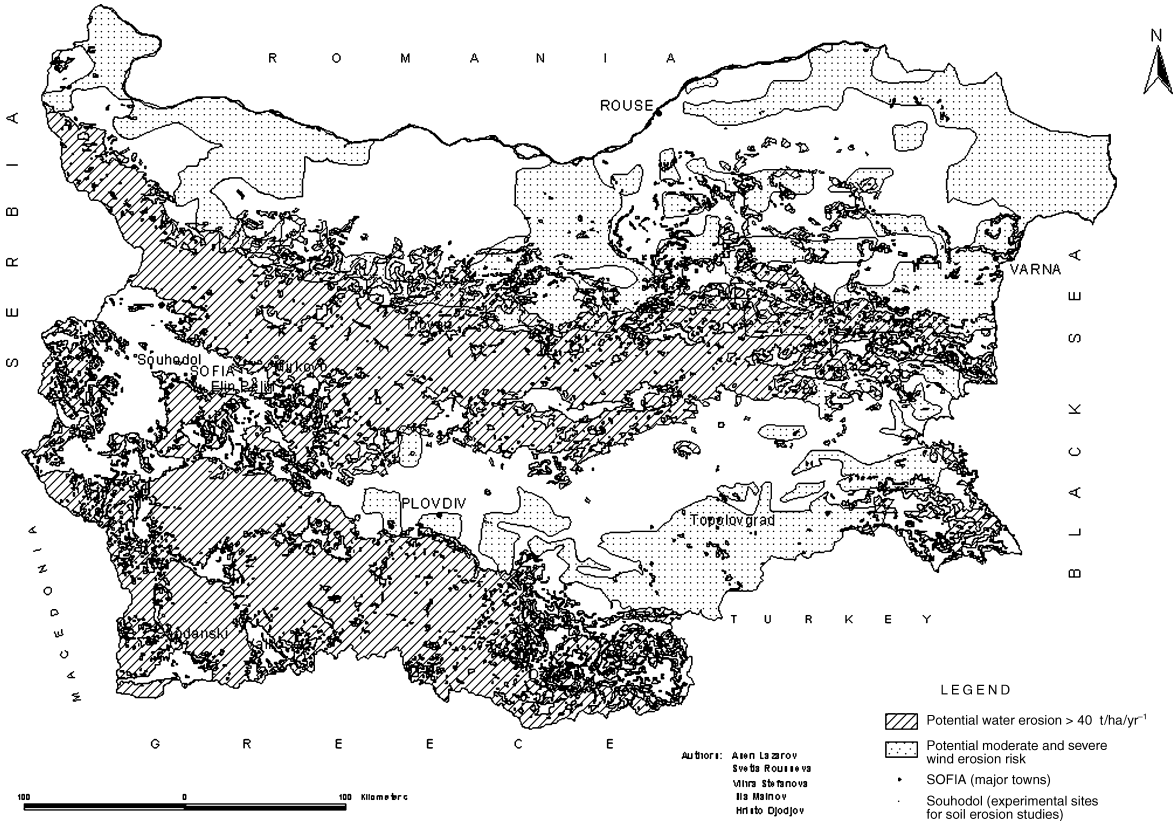

Figure 1.14.1 Erosion map of Bulgaria

mountains'), the Sub-Balkan Valleys, the Sredna Gora Mountains, the central Thracian Plain and the Rhodope Mountains, along the border with Greece. The easternmost sections near the Black Sea are hilly and they gradually gain height to the west until the westernmost part of the country is entirely high ground. At the western end of the Rhodope Mountains, in south-western Bulgaria, are the Pirin Mountains and the Rila Massif, which culminates in Musala Peak (2925 m), the highest point in the Balkans. Several smaller ranges lie along the western boundaries. The oldest in terms of geological age are the Moezian Platform (the Danubian Hilly Plain), the Rila Massif and the Rhodopes. The mountains of Stara Planina and Sredna Gora, and the Sub-Balkan Valleys date from a later geological age.

The average altitude of Bulgaria is 470 m above sea level. The lowlands (up to 200 m) make up 31% of the territory, the plains and the low hills (200–600 m) 41%, the low mountains (600–1000 m) 15%, the medium height mountains (1000–1600 m) 10% and the high mountains (over 1600 m) 3%. Over two-thirds of the land slopes above 8%, i.e. rolling to hilly landscapes prevail. Steep lands with slope gradients of 8–30% occupy 52% of the territory and 16% of the land has very steep slopes exceeding 30%.

The Balkan Mountains divide Bulgaria into two nearly equal drainage systems. The larger system drains northwards to the Black Sea, mainly by way of the Danube River. This system includes the entire Danubian Plateau and a stretch of land running 48–80 km inland from the coastline. The second system drains the Thracian Plain and most of the higher lands of the south and southwest to the Aegean Sea.

Geographical position and varied relief set the pattern for the spatial distribution of the climate parameters. For the larger part of the country the climate is moderately continental with four clearly differentiated seasons. The country's average annual temperature is 10.5 °C with a range over the country's territory of 9.9 °C. The coldest month is January (average temperature –1.0 °C with a range of 6.9 °C) and the warmest is July (average temperature 21.0 °C with a range of 12.3 °C). The annual mean total precipitation in Bulgaria is approximately 500–650 mm, with variations ranging from 440 to 1020 mm. The highest monthly values are in June and May, with a mean total varying between 55 and 85 mm. February and sometimes March and September are the driest months with mean totals varying between 30 and 45 mm. Mean precipitation during the warm months, i.e. April–September, is 333 mm with a standard deviation of 72 mm (Alexandrov, 2000). The rainfall erosivity index (EI_{30}) is in the range 600–1000 MJ mm ha^{-1} h^{-1} for 51.2% of the territory and it exceeds 1000 MJ mm ha^{-1} h^{-1} for 12.3% of the country (Rousseva, 2002).

Results from recent studies on the annual fluctuations of the main meteorological elements indicate a trend towards warming accompanied by an increase in evaporation losses, precipitation and river flow in autumn and winter and a river flow decrease in summer. There is an increase in winter precipitation and a decrease in summer precipitation in southern Bulgaria, and an increase in summer precipitation in the northern part of the country (Alexandrov, 2002; Slavov, 2002).

Wind direction, velocity and frequency distribution depend on the season and the topography. North-west and west winds cause warming in spring and bring rainfall in summer. Winds from the north-east bring dry and cold continental air in winter. On average, there are 3–5 windy days with a dominant wind velocity of 5–10 m s^{-1} and 1–2 days with a wind velocity of 11–15 m s^{-1} per month during the spring. Strong spring dry spells set in every third year.

The structure of the soil cover of Bulgaria is very complicated and often does not reflect the present climate and vegetation conditions. Five types of pedo-climate can be recognized in the country: Crio-Udic, Meso-Udic, Meso-Ustic, Meso-Xeric and Thermo-Xeric (Boyadgiev, 1994b). There are four soil regions: (i) Cambisol–Podzol–Leptosol region with Luvisols; (ii) Chernozem–Kastanozem–Phaeozem region with Luvisols; (iii) Luvisol region with Leptosols and Planosols; and (iv) Vertisol region of Central Bulgaria. The soil map of Bulgaria shows a mosaic pattern of great variability of soils and 20 out of the 30 WRB soil reference groups can be found. The most widely distributed are Chernozems, occupying about 29% of the country, followed by Luvisols 20%, Cambisols 16.5%, Planosols 15% and Vertisols 7% and 12.5% of the territory is covered by Fluvisols and other soils (Boyadgiev, 1994c). The dominant agricultural soils have low organic matter content and an unstable structure that determines their high erodibility. Values of the soil erodibility factor (K) vary from 0.003 to 0.055 t ha h ha^{-1} MJ^{-1} mm^{-1} owing to the diversity of the soils and their large spatial variability (Rousseva, 2002).

1.14.3 PERMANENT LAND COVER

Agricultural land covers 56.3% of Bulgaria, forest 35.3% and settlements, industries, transport and infrastructure 6.7%; water bodies occupy 1.8% of the territory. Cropland covers 39.8% of the total area, rangeland and pastures 11.6% and permanent crops 1.9% (Kostov, 2001).

Mostly because of severe erosion, the area of arable land in Bulgaria decreased from 4 880 900 to 4 642 700 ha at an average annual rate of about 8000 ha yr^{-1} from 1960 to 1990 (Rousseva *et al.*, 1992). Since 1990, the area of cropland has grown from 3 847 800 to 4 424 000 ha but the cropped area has dropped significantly, while the abandoned field crop area, mostly lands sloping at over 6 %, has increased to 1 502 000 ha (Ivanova *et al.*, 1993; Penevska *et al.*, 1996; Kostov, 2001). The ratio of cereals (mostly wheat, barley and oats) to row crops (mostly maize, sunflower, sugar beets, potatoes and tobacco) has varied

from 1.2 to 1.6 during the period 1989–2000 (Ivanova *et al*., 1993; Penevska *et al*., 1996; Kostov, 2001). The total area of the forest land is 3 914 355 ha and 86.2% of it is afforested.

1.14.4 LAND MANAGEMENT

Until the end of World War II, most landowners in Bulgaria were in possession of small-sized properties: more than 50% of the area of agricultural land (over 2.9×10^6 ha) was represented by fields smaller than 10 ha. There was a mosaic of small-sized private fields with different crops. The boundary strips, being covered with grass, trees and bushes, most often followed the slope contours and thereby protected the soil from erosion.

In the late 1940s and the 1950s, during the period of establishment of cooperative farms, fields were reconstructed to accommodate large-scale cultivation and the naturally established field boundary strips that delimited the former properties were removed. As a result, the natural habitats of many biological species were lost and the corridors connecting agricultural and forest territories broken. The specialization and concentration of agricultural production led to a loss of biodiversity, reducing the stability of agroecosystems and resulting in accelerated land degradation. Thus, water and wind erosion became significant soil degradation processes owing to irrational land management. The processes of land degradation were also accelerated because the communist ideology that land, being a natural resource, cannot be held privately but only as a public property meant that everybody could exploit the land but none was obliged to take care of it.

Since 1990, there has been a radical change in the political system in Bulgaria, followed by desirable changes in the economy. This period can be characterized by (i) privatization of agricultural land, (ii) market-oriented production and (iii) steps towards European integration, with special attention to quality standards and environmental aspects. At present, most agricultural land is on lease to small cooperatives or larger individual land users. A few resources for soil protection come from pre-accession funds, such as the Special Accession Programme for Agriculture and Rural Development (SAPARD).

1.14.5 HISTORICAL EVIDENCE FOR EROSION

Although evidence of inhabitation of Bulgaria exists from the Paleolithic (Montana region) and the Copper Age (Varna region), the population has been growing and increasing its pressure on the environment only since the Thracian ethnic community began to develop by the middle of the second millennium BC. Prolonged agricultural use of the land and clearance of a large part of the forest, which started in 15–16th centuries, resulted in eroded soils with soil profile truncation occupying 11.8% of the territory and another 41.4% is covered by shallow soils (Boyadgiev, 1994a).

As a result of irrational land management in the 1950s, about 10% of arable land became completely eroded and not suitable for cropping, or even afforestation by the end of the 1960s and 1970s. The dam siltation rates were as much as 5–6 times higher than expectations. Some small ponds built by cooperative farms completely silted up in 2–3 years. About 20% of field crops were swept away periodically by dust storms. Rainstorms formed deep rills and gullies on sloping land and muddy torrents damaged field crops, highways, railroads, bridges and residential and utility buildings.

1.14.6 CURRENT EROSION PROCESSES

Erosion is recognized as the major soil degradation process in Bulgaria. Three types of soil erosion are identified depending on the driving force – water, wind and irrigation.

1.14.6.1 Water Erosion

Field plots have been set up in experimental sites of the N Poushkarov Institute of Soil Science, the Complex Experimental Station of Kardzhali, the Institute for Mountain Agriculture and Stock-breeding and the Institute for Barley to study soil erosion in diverse soil, climate, topography, cover and management conditions on agricultural land since 1958. The information obtained has been used as a base for evaluation of the soil erosion processes and factors, design, validation and use of models predicting soil loss due to water erosion, soil conservation planning and optimization of measures for soil erosion control (Biolchev *et al*., 1977; Onchev, 1983; Rousseva, 2002). The Forest Research Institute has organized four experimental stations and observation points for studying erosion processes in the forest lands at small experimental watersheds (7.5–64.8 ha) and a large amount of data has been collected on the water-regulating and soil-protecting effects of coniferous plantations, rainfall, interception, surface runoff and water discharge (Kerenski, 1972; Angelov, 1986; Mandev, 1984, 1995, 1996; Marinov, 1995). Selected data from soil erosion studies are presented in the following tables.

The data presented in Table 1.14.1 show that the experimental sites represent very different climatic conditions. The average annual number of erosive rains varies between the sites from one to seven and the respective amount of a single event from 15 to 24 mm. The annual kinetic energies vary from 7.7 to 29.3 MJ ha^{-1} and the rainfall erosivity indices from 204 to 796 MJ mm ha^{-1} h^{-1}. In addition to the high spatial variability, rainfall parameters also vary from year to year. The mean standard error of the average annual number of erosive events is 0.8, varying from 0.3 (Sandanski) to 1.3 (Troyan and Valkosel). The mean standard errors of the mean amount and the mean intensity of a single rainstorm event are 2.1 mm and 2.1 mm h^{-1}, respectively, varying from 0.7 (Mirkovo) to 4.5 mm (Dzhebel) and from 0.7 (Souhodol) to 5.0 mm h^{-1}

TABLE 1.14.1 Measured average annual values of the main characteristics of erosive rainfalls and standard deviations for nine experimental sites in Bulgaria

				For single event		Annual sums	
Experimental site	Value	Period of measurement	Number of rainfalls	Mean amount (mm)	Mean intensity (mm h^{-1})	*KE* (MJ ha^{-1})	EI_{30} (MJ mm ha^{-1} h^{-1})
Dzhebel	Mean	04/1980–	6.8	19.9	9.5	16.5	390.6
	SD	10/1986	2.8	11.9	9.2	8.3	249.5
Elin Pelin	Mean	04/1978–	2.1	18.0	21.6	9.6	374.7
	SD	10/2002	1.8	9.3	9.3	9.6	517.3
Mirkovo	Mean	04/1959–	6.8	20.3	9.7	29.0	796.0
	SD	10/2000	3.7	4.6	6.4	16.4	571.2
Rouse	Mean	04/1981	5.3	19.5	12.5	24.1	765.7
	SD	10/2000	3.2	4.6	9.0	16.4	568.8
Sandanski	Mean	04/1990–	1.4	24.0	18.9	8.3	371.1
	SD	10/2002	1.1	13.4	6.4	6.6	356.5
Souhodol	Mean	04/1971–	6.0	21.6	5.3	26.2	567.5
	SD	10/2000[a]	3.1	9.3	2.8	14.4	586.3
Topolovgrad	Mean	04/1986–	4.5	15.1	5.5	29.3	559.2
	SD	10/2000	2.1	9.5	6.3	18.0	513.2
Troyan	Mean	04/1984	5.5	18.1	4.5	20.3	557.0
	SD	10/1994	4.2	3.0	4.2	20.4	811.3
Valkosel	Mean	04/1985–	5.6	15.5	24.4	7.7	204.4
	SD	10/1989	2.9	2.5	11.2	4.1	119.3

[a]Excluding the periods 1979–80 and 1987–97.

TABLE 1.14.2 Measured average annual values and standard deviations of eroded soil from runoff plots on 21 bare soils at nine experimental sites in Bulgaria

Soil	Town	Period of measurement	Plot length (m)	Plot slope (%)	Eroded soil ($t\,ha^{-1}\,yr^{-1}$)	
					Mean	SD
Haplic Kastanozem 1	Rouse	04/1981–10/1999	10	12.3	34.17	38.63
Haplic Kastanozem 2	Rouse	04/1981–10/1986	8	11.4	15.56	12.66
Haplic Kastanozem 3	Rouse	04/1981–10/1986	8	18.6	42.53	32.54
Chromic Luvisol 1	Dzhebel	04/1980–10/1986	8	12.3	6.59	5.12
Chromic Luvisol 2	Dzhebel	04/1980–10/1986	8	12.3	0.82	0.62
Chromic Luvisol 3	Dzhebel	04/1980–10/1986	8	12.3	2.84	2.12
Chromic Luvisol 4	Dzhebel	04/1980–10/1986	8	12.3	1.73	1.51
Chromic Luvisol 5	Dzhebel	04/1980–10/1986	8	12.3	1.22	1.02
Chromic Luvisol 6	Dzhebel	04/1980–10/1986	8	12.3	10.15	1.70
Chromic Luvisol 7	Dzhebel	04/1980–10/1986	8	12.3	10.83	1.62
Chromic Luvisol 8	Souhodol	04/1981–10/1986	8	9.5	3.48	3.55
Chromic Luvisol 9	Souhodol	04/1981–10/1986	8	10.5	6.01	5.71
Chromic Luvisol 10	Topolovgrad	04/1986–10/1991	10	26.6	17.56	9.15
Chromic Luvisol 11	Topolovgrad	04/1992–10/1998	10	10.5	6.10	10.68
Chromic Luvisol 12	Topolovgrad	04/1992–10/1998	10	15.9	9.92	14.23
Chromic Luvisol 13	Elin Pelin	04/1977–10/1995	30	17	9.94	22.49
Chromic Luvisol 14	Sandanski	04/1973–10/1993	40	25	75.41	53.28
Dystric Cambisol 1	Mirkovo	04/1960–10/1970	70	14.5	17.79	17.83
Dystric Cambisol 2	Mirkovo	04/1985–10/1988	70	14.5	42.84	20.94
Dystric Regosol	Valkosel	04/1985–10/1989	8	11	1.65	1.60
Distric Planosol	Troyan	04/1989–10/1994	4	10.0	3.72	3.71

(Valkosel), respectively. Corresponding values for the average annual kinetic energy are 3.2, 1.8 (Sandanski and Valkosel) and 6.2 $MJ\,ha^{-1}$ (Troyan) and for the rainfall erosivity index 37.6, 8.8 (Mirkovo) and 103.5 $MJ\,mm\,ha^{-1}\,h^{-1}$ (Elin Pelin).

The data presented in Table 1.14.2 show the range of soil loss from 21 soils located at nine experimental sites, measured for periods from 4 to 21 years. The slope of the plots ranged from 9.5 to 26.6% and the lengths were 4, 8, 10, 30, 40 and 70 m. The mean average annual soil loss of the studied soils is 15.3 $t\,ha^{-1}\,yr^{-1}$. Average annual soil losses from Haplik Kastanozem ranged from 15.6 to 42.5 $t\,ha^{-1}\,yr^{-1}$ and those from Chromic Luvisols from 0.8 to 75.4 $t\,ha^{-1}\,yr^{-1}$. The discussed average annual values are characterized by very high temporal variability. The average annual soil loss from Dystic Cambisol measured from 1960 to 1970 was 17.8 $t\,ha^{-1}\,yr^{-1}$ with a standard error of 5.4 $t\,ha^{-1}\,yr^{-1}$, but 42.8 $t\,ha^{-1}\,yr^{-1}$ and a standard error of 10.5 $t\,ha^{-1}\,yr^{-1}$ when measured between 1985 and 1988. The mean standard error of the soil losses included in the data set is 4 $t\,ha^{-1}\,yr^{-1}$, varying from soil to soil from 0.2 (Chromic Luvisol 2) to 13.3 $t\,ha^{-1}\,yr^{-1}$ (Haplic Kastanozem 3).

Table 1.14.3 shows selected data (Kroumov and Malinov, 1989; Kroumov, 1995; Rousseva *et al*., 2004) from four experimental sites for soil erosion studies in agricultural lands to illustrate the range of soil loss from major crops. The mean soil loss rate for cover crops (wheat, triticale, rye, lucerne, perco, grass mixture and brassica) is 0.8 $t\,ha^{-1}\,yr^{-1}$ and that for row crops (maize, sunflower and oriental tobacco) is 2.5 $t\,ha^{-1}\,yr^{-1}$. The differences between the soil losses observed for a certain crop (wheat or maize) grown in different locations demonstrate well the influence of climate and soil factors on soil erosion rates. Standard deviations indicate temporal variability of erosion rates. The mean standard error of the erosion rates is 0.6 $t\,ha^{-1}\,yr^{-1}$,

TABLE 1.14.3 Measured average annual values and standard deviations of eroded soil from major field crops observed at four experimental sites in Bulgaria

Crop	Town	Period of measurement	Plot length (m)	Plot slope (%)	Eroded soil ($t\,ha^{-1}\,yr^{-1}$)	
					Mean	SD
Oriental tobacco	Valkosel	04/1985–10/1989	8	11	0.70	0.56
Perco[a]	Valkosel	04/1985–10/1989	8	11	0.30	0.06
Triticale	Valkosel	04/1985–10/1989	8	11	0.33	0.18
Rye	Valkosel	04/1985–10/1989	8	11	0.27	0.15
Wheat	Rouse	04/1997–10/2002	5	12	0.27	0.26
Maize	Rouse	04/1997–10/2002	5	12	1.78	2.24
Lucerne	Rouse	04/1997–10/2002	5	12	1.01	2.17
Wheat	Souhodol	04/1998–10/2002	8	10	0.76	0.83
Maize	Souhodol	04/1998–10/2002	8	10	1.50	1.00
Brassica	Souhodol	04/1998–10/2002	8	10	0.42	0.41
Wheat	Topolovgrad	04/1998–10/2002	10	11	1.94	1.55
Maize	Topolovgrad	04/1998–10/2002	10	11	5.15	3.70
Sunflower	Topolovgrad	04/1998–10/2002	10	11	3.59	3.01
Herbaceous mixture	Topolovgrad	04/1998–10/2002	10	11	1.96	1.40

[a]Hybrid of Chinese cabbage and winter rape.

varying from 0.03 for Perco in Valkosel to $1.7\,t\,ha^{-1}\,yr^{-1}$ for maize in Topolovgrad. The high temporal variability of the soil erosion rates observed in Rouse and Topolovgrad is associated with the increased rainfall variability in those regions since 1990s.

The data in Table 1.14.4 show the range of soil loss measured on forest land. The average annual amount of soil eroded from grass lands varies from $0.03\,t\,ha^{-1}\,yr^{-1}$ with a standard error of $0.01\,t\,ha^{-1}\,yr^{-1}$ to $6\,t\,ha^{-1}\,yr^{-1}$ with a standard error of $2\,t\,ha^{-1}\,yr^{-1}$, depending on the type of grass, the plot's topography and the experimental site. The observed average annual soil loss from Scots pine plantations is about $0.03\,t\,ha^{-1}\,yr^{-1}$ with a standard error of $0.01\,t\,ha^{-1}\,yr^{-1}$ and that from thin Oak forest is $0.47\,t\,ha^{-1}\,yr^{-1}$ with a standard error of $0.16\,t\,ha^{-1}\,yr^{-1}$.

TABLE 1.14.4 Measured average annual values and standard deviations of eroded soil observed at two experimental sites

Crop	Town	Period of measurement	Plot length (m)	Plot slope (%)	Eroded soil ($t\,ha^{-1}\,yr^{-1}$)	
					Mean	SD
Grasslands	Elin Pelin	04/1975–10/1995	30	17	0.034	0.044
Scots pine plantation	Elin Pelin	04/1974–10/1995	30	17	0.029	0.039
Grasslands, meadow >0.65 cover (northern aspect)	Sandanski	04/1969–10/1993	60	24	0.455	1.014
Grasslands (after fallow, southern aspect)	Sandanski	04/1973–10/1993	40	25	5.980	9.360
Thin oak forest	Sandanski	04/1969–10/1993	40	32	0.468	0.806
Coniferous forest plantations (Scots pine; Austrian black pine)	Sandanski	04/1969–10/1993	50	32	0.026	0.065

A number of studies have been carried out to investigate the effects of different measures to control soil erosion, e.g. (i) basic tillage – mouldboard ploughing, subsurface and no-tillage; (ii) spring pre-sowing tillage operations; (iii) crop rotations; and (iv) intermediate crops have been studied on Haplic Kastanozem in Rouse and on Chromic Luvisol in Topolovgrad (Tzvetkova *et al*., 1994; Bakalov, 1995). The average annual soil loss on maize grown on no-tilled Haplic Kastanozem during the period 1989–91 was 2.20 t ha^{-1} yr^{-1}; with subsurface ploughing it was 2.21 t ha^{-1} yr^{-1}, whereas from mouldboard ploughing along the slope it was more than three times higher (7.48 t ha^{-1} yr^{-1}). The average annual soil losses on sunflowers grown on Chromic Luvisol during the period 1987–90 were 1.42, 1.12 and 4.05 t ha^{-1} yr^{-1} for no-tillage, subsurface ploughing and mouldboard ploughing along the slope respectively. The crop yields on no-tillage, however, were lower on both soil types. Triticale + winter peas on Haplic Kastanozem and winter vetch + barley on Chromic Luvisol in the period 1989–92 increased crop productivity and better protected the soil from erosion.

During the period 1988–92, Vateva *et al*. (2003) studied the effects of mineral fertilization on soil erosion from severely degraded rangeland with a slope of 12–15° at the experimental site near Topolovgrad. The mean annual soil erosion rate established for the unfertilized treatment was 1.46 t ha^{-1} yr^{-1} with standard error 0.21 t ha^{-1} yr^{-1}. The soil erosion rates decreased with increase in the fertilization rate and at $N_{180}P_{180}K_{60}$ it was as much as four times lower than that at $N_0P_0K_0$.

The data in Table 1.14.5 demonstrate the soil loss reduction effect of grazing control, fertilization, and clearing bushes and stones to improve degraded pasturelands in the area of Valkosel. The data in Table 1.14.6 illustrate the soil conservation effects of grass and forest strips. The soil losses from the respective bare soils are shown in Table 1.14.2.

The data in Table 1.14.7 show that the average annual sediment load of the major Bulgarian rivers varies by over two orders of magnitude, from about 0.1 t ha^{-1} yr^{-1} for the Maritsa (at Pazardzhik) and the Struma (at Razhdavitsa) to 13.2 t ha^{-1} yr^{-1} for the Arda at Dzhebel. Downstream sediment yield estimates show a fourfold increase (from 0.109 to 0.436 t ha^{-1} yr^{-1}) for the Maritsa River between Pazardzhik and Harmanli, a fivefold increase (from 0.310 to 1.547 t ha^{-1} yr^{-1}) for the Tundzha River between Pavel Banya and the Elhovo and a ninefold increase (from 0.115 to 1.009 t ha^{-1} yr^{-1}) for the Struma River between Razhdavitsa and Malo Pole.

Lazarov *et al*. (2002) developed a Geographic Information System for assessing the risk of sheet erosion and estimated potential erosion risk exceeding 100 t ha^{-1} yr^{-1} for 10.4% of the country's territory, from 40 to 100 t ha^{-1} yr^{-1} for 19.5%, from 10 to 40 t ha^{-1} yr^{-1} for 31.7% and less than 20 t ha^{-1} yr^{-1} for 25.9%. The estimated 'actual' average annual soil loss rates ranged from 0.14 t ha^{-1} yr^{-1} on forestlands to 2.69 t ha^{-1} yr^{-1}

TABLE 1.14.5 Measured average annual values and standard deviations of eroded soil from natural pasture at the experimental site of Valkosel

Land use	Period of measurement	Plot length (m)	Plot slope (%)	Eroded soil (t ha^{-1} yr^{-1})	
				Mean	SD
Natural pasture, no grazing					
No improvements	04/1984–10/1987	8	11	0.065	0.14
$N_{100}P_{100}K_{100}$	04/1984–10/1987	8	11	0.070	0.15
Clearing bushes and stones	04/1984–10/1987	8	11	0.056	0.12
$N_{10}P_{10}K_{10}$ + clearing brushes and stones	04/1984–10/1987	8	11	0.025	0.04
Natural pasture no control grazing					
No improvements	04/1984–10/1987	8	11	0.516	0.96
$N_{100}P_{100}K_{100}$	04/1984–10/1987	8	11	0.062	0.11

After Kroumov and Malinov (1989).

TABLE 1.14.6 Measured average annual values and standard deviations of eroded soil from sloping land with grass and forest strips at the experimental site of Mirkovo

Land use	Period of measurement	Plot length(m)	Plot slope (%)	Eroded soil ($t\ ha^{-1}\ yr^{-1}$)	
				Mean	SD
Fallow 70 m + grass strip 5 m	04/1960–10/1970	75	14.5	5.13	9.60
Fallow 70 m + grass strip 10 m	04/1960–10/1970	80	14.5	4.59	4.47
Fallow 70 m + grass strip 20 m	04/1960–10/1970	90	14.5	1.01	2.51
Fallow 70 m + forest strip 10 m	04/1960–10/1970	80	14.5	2.37	5.82
Fallow 70 m + forest strip 20 m	04/1960–10/1970	90	14.5	0.79	1.75
Fallow 70 m + forest strip 40 m	04/1960–101970	110	14.5	0.26	0.88
Fallow 70 m + grass strip 5 m	04/1985–10/1988	75	14.5	1.20	1.19
Fallow 70 m + grass strip 10 m	04/1985–10/1988	80	14.5	0.68	0.67
Fallow 70 m + grass strip 20 m	04/1985–10/1988	90	14.5	0.21	0.21
Fallow 70 m + forest strip 10 m	04/1985–10/1988	80	14.5	0.25	0.26
Fallow 70 m + forest strip 20 m	04/1985–10/1988	90	14.5	0.12	0.13
Fallow 70 m + forest strip 40 m	04/1985–10/1988	110	14.5	0.001	0.0003

After Biolchev (1975) and Malinov (1999).

TABLE 1.14.7 Average annual sediment yields in the major Bulgarian rivers

River	Gauge station/town	Period of measurement	Sediment yield ($t\ ha^{-1}\ yr^{-1}$)
Nishava	223/Kalotina	1980–1989	0.272
Lom	88/Traikovo	1980–1989	0.309
Ogosta	121/Miziya	1980–1989	0.388
Skat	120/Miziya	1980–1989	0.157
Iskar	118/Oryahovitsa	1980–1989	0.447
Yantra	82/Karanci	1980–1989	0.934
Rusenski Lom	1/Besarabovo	1963–1977	0.877
Kamchiya	11/Grozdevo	1980–1989	0.574
Luda Kamchiya	10/Asparuhovo	1982–1972	3.869
Maritsa	252/Pazardzhik	1980–1989	0.109
Maritsa	30/Plovdiv	1982–1990	0.285
Maritsa	307/Harmanli	1980–1989	0.436
Luda Yana	251/Sbor	1975–1984	0.637
Topolnitsa	240/Poibrene	1980–1989	1.303
Stryama	325/Banya	1980–1989	0.202
Arda	312/Dzhebel	1980–1989	13.218
Varbitsa	312/Dzhebel	1980–1989	3.376
Tundzha	338/Pavel Banya	1980–1989	0.315
Tundzha	373/Elhovo	1980–1989	1.546
Struma	201/Razhdavitsa	1980–1989	0.115
Struma	220/Malo Pole	1980–1989	1.009
Dzherman	187/Dupnitsa	1980–1989	1.241

After Gergov and Fitova (1995).

on rangeland and from 4.76 t ha^{-1} yr^{-1} on cropland to 12.65 t ha^{-1} yr^{-1} on vineyards and orchards, resulting in a net average annual soil loss estimate of 32 Mt yr^{-1}. Figure 1.14.1 shows the areas with potential erosion risk exceeding 40 t ha^{-1} yr^{-1}.

1.14.6.2 Wind Erosion

Wind erosion occurs on flat plains and deforested regions. The area of land with wind erosion risk is assessed at 1 657 386 ha (29% of cultivated lands) and the resulting estimated annual soil loss can vary from 30 to 60 Mt yr^{-1} depending on the intensity of the dust storms (Djodjov, 1982; Peev, 1989; Georgiev *et al*., 1995; Djodjov *et al*., 1997). The areas at high wind erosion risk are associated with low forest density and high soil susceptibility to wind erosion (Figure 1.14.1). It has been recognized that successful protection of soils from wind erosion requires application of modern methods for evaluating the driving forces and defining the areas under wind erosion risk. Malinov and Djodjov (1995, 2003) found that soils with high wind erosion susceptibility cover about 10% of the country's territory, with moderate susceptibility 24.8%, and with low susceptibility 11%. Stoev *et al*. (1997) and Djodjov *et al*. (2003) showed that the wind erosion risk is high on 9946 km^2 (9% of the country's territory), moderate on 19 934 km^2 (18%) and low on 33 708 km^2 (30.4%). The regions most affected by wind erosion are Sliven, Pleven, Montana, Veliko Tarnovo, Razgrad and Vratsa. The development of a Geographic Information System for wind erosion risk assessments is under consideration.

1.14.6.3 Irrigation Erosion

The risk of irrigation erosion is negligible as far as it impacts on the irrigated arable land sloping over 3°, most of which has been abandoned since 1990. The area of irrigated land in Bulgaria was about 1 000 000 ha (25% of the arable land) until 1990. It has been recognized that the soil erosion risk is highest with furrow irrigation, varying from 2.5 to 8.6 t ha^{-1} for one water application (Krasteva, 1983), and that the use of artificial rain and drip irrigation reduces the soil losses significantly. However, most of the irrigation systems have been abandoned since 1990.

1.14.6.4 Landslide, Coastal and River Erosion

Landslides, coastal erosion and riverbank erosion are very common in Bulgaria. The highest concentration of landslides is encountered along the Black Sea coast, the high riverbank of the Danube, northern Bulgaria, southwest Bulgaria and the Rhodope Mountain. At present, 960 landslides have been registered in 350 settlements, resorts and residential areas, covering a total area of 22 000 ha (Petrov, 2002). Another 250 landslides affect the national road network. Past and contemporary landslides of a volume measured in billions of cubic metres affect about 150 km of the Bulgarian Danube riverbank and also the right-side river banks of the Danube tributaries Skomlya, Lom, Tsibritsa, Ogosta, Iskar, Vit and Yantra. The length of the reinforced banks of the Danube is 59 km and riverbank erosion is active over 48.5 km of the bank, and another 50.2 km of it is at high erosion risk (Petrov, 2002).

1.14.7 SOIL CONSERVATION MEASURES

Soil conservation measures, such as contour cropping and drawing drainage furrows after sowing, have been practised on Bulgarian sloping lands from time immemorial. There are still existing old stone-reinforced terraces and field boundary strips overgrown with bushes and trees in the mountainous and semi-mountainous regions, where vines and tobacco have been grown for generations.

Organised erosion control in Bulgaria has a history of over 95 years. The first Section (Bureau) of Torrent Stabilization and Afforestation was established in 1905. Significant erosion control activities have been performed in the forest area and on the hydrographic network with very good results (Kerenski and Marinov, 1995; Kostov *et al*., 1995; Dobrev *et al*., 1998; Zakov and Marinov, 2003). Examples exist of very successful activities for controlling torrents and for stabilizing torrent beds and landslides (Panov, 2000). During the period 1905–51 erosion control activities were directed by the 'Sections for stabilizing of torrents and forestation', of which there were 56 at the end of this period. Since 1951, these activities became the responsibility of the State Forestry Services. During the last few years, private companies performed these activities. An area of 44 700 ha was afforested and 88 000 m^3 of stone barrages were constructed before 1931 and the total area of eroded lands afforested before 1951 was 170 000 ha. The amount of barrage construction during that period was about 130 000 m^3.

In the early 1970s, soil erosion was recognized as a national problem of primary significance owing to the damage it caused to the national economy. Government decrees, aimed at organizing soil conservation and landslide control, were issued at that time. Research teams were formed to study soil erosion processes and to put forward efficient erosion control in agricultural and forest lands. Bureaus to organize erosion control projects and Melioration and Erosion Control Enterprises to fulfil those projects were established in the main regional towns of the country. The Bureaus collaborated with researchers designed the National Long-term Erosion Control Programme (NLECP) that recommended erosion prevention measures based on land capability evaluation and the estimated average annual soil loss rates. The NLECP made provisions for the design of erosion control measures at the level of catchments, administrative territorial units or the area of the cooperative farms.

During the late 1970s and early 1980s, many erosion control projects for particular watersheds and cooperative farms were developed. However, those projects have never been fulfilled completely owing to neglect by the specialized enterprises that were to enact them in practice. For instance, erosion control measures such as partitioning of large fields and their orientation with shorter sides along the slope, strip-cropping, buffer strip-cropping, soil-protecting crop rotations and conservation tillage practices were neglected. Thereby, the State budget funding granted for soil erosion control was used mainly for hydro-technical reinforcement of torrents, terracing, pasture land improvements (fencing, clearing from stones and bushes, filling up the small gullies, constructions of collection ditches, sod improving, fertilization, etc.), river-bed corrections, constructions of small water reservoirs, etc. About 326 500 m^3 of barrages, 329 000 m^3 of stone sills and 248 000 m of bank hedges were constructed during the period 1952–80. This period is also remarkable for mass afforestation of 486 200 ha, reaching 80 000 ha in some years, and the development of 20 327 ha of shelterbelts. The stabilization of the torrents was recognized as a substantial part of erosion control activities. The stabilization of the bed of the torrential Perperek River, in the vicinity of Kardjali, is an example of successful biological stabilization. It has resulted in retention of large amounts of sediments outside the dam and provision of a considerable area of land suitable for agricultural production. An important part of erosion control activities took place in dam watersheds. More than 80 large complex erosion control projects have been designed and applied in dam and torrent watersheds. The measures led to limited dam siltation. Georgiev (1993) showed that the coefficient of siltation, defined as the ratio of actual to predicted siltation, was low for nine of 15 dams studied, the deposition was within the range of acceptable values for two dams, with high values in four dams. According to a report of the Ministry of Agriculture in 1987, the funds for the implementation of the NLECP had protected about 20% of the agricultural land with high erosion risk.

The 1990s were characterized by a transition towards a market-oriented economy and land reform, which assumes mixed land ownership, decreased size of farming units and agricultural fields and restored ownership. These conditions should have allowed the establishment of a flexible, environmentally friendly and sustainable agriculture, but in reality that potential has not been met. Considering erosion control on agricultural land, the

1990s are marked as a decade of complete neglect. Permanent constructions to control erosion, once completed, have not been maintained so their disintegration has been in progress. Many terraces have been damaged, collection ditches have been broken and grassed land has not been protected from excessive grazing. The number of Soil Conservation Designing Bureaus has been reduced and they changed the field of their activity. The Specialized Erosion Control Enterprises have neglected soil conservation construction works. Some decrease in afforestation works has taken place in the 1990s and especially since 1995, when the mean annual erosion control afforestation rate was below $600\,\mathrm{ha\,yr^{-1}}$. The erosion control hydro-technical construction work rate has also decreased significantly, while barrages of about $1000\,\mathrm{m^3\,yr^{-1}}$ have been built. Two national programmes for landslides and riverbank erosion have been implemented since 1998 at a total cost of €48 million. These programmes were aimed at stabilizing (i) 59 landslides along the Black Sea coast at a total cost of €35 million and (ii) 48.5 km of the Danube riverbank affected by bank erosion at a total cost of €13 million. A regional programme for stabilization of landslides along the high Danube bank is under consideration.

1.14.8 SOIL CONSERVATION POLICY

There is no overall strategy and policy to guarantee efficient protection of the soils as a natural resource. The soil-protection legislation is incomplete. Separate provisions can be found in several regulative acts and in the Law on Protection of the Agricultural Lands, but they are insufficient to ensure land protection from erosion degradation.

The Ministry of Agriculture and Forestry has the responsibility for developing the policy for the use and protection of agricultural land and forest. The Ministry of Environment and Water is responsible for the prevention of pollution and protection of the land as a natural resource. The Ministry of Regional Development and Public Works and the enterprise 'Geozashtita – EOOD' are the management bodies fulfilling geological control activities, including monitoring and control of landslides, marine abrasion and riverbank erosion.

During the period 1955–89, soil erosion control in Bulgaria was based on Government and State decrees, which resulted in the development of the NLECP. All the activities foreseen by the NLECP have been suspended since 1989. Since 1990, soil erosion prevention has been a subject of legislation, approved by the National Assembly. There are two laws concerning the conservation of agricultural soils:

- The Law on Land Ownership and the Use of Agricultural Land was issued in 1991 (Anon., 1991a) and it was expanded and changed more than 20 times before 2000. It enacts that '... the landowners and the land users are obliged to preserve the land from erosion and to enable implementation of erosion control measures funded by the State...'. According to this law, erosion prevention is financed by a national fund: 'Prevention and improvement of the productivity of agricultural lands'. That fund accumulates resources from rents and sales of state owned land, taxes and different sanctions.
- The Law on Protection of Agricultural Land was issued in 1996 (Anon., 1996a) and it was changed five times before 2000. It regulates the issues of protection of agricultural land from damage, the recovery and improvement of soil fertility and determines the conditions and the order for changes of the type of landuse. Concerning soil erosion, the law makes the following provisions:

  – the landowners and the landusers are obliged to preserve the land from erosion;
  – the Ministry of Agriculture and Forestry makes provisions for official information on the land quality and on the potential risks for its deterioration because of erosion;

–the Ministry of Agriculture and Forestry maintains information systems on agricultural soil resources, which include a special register for the lands with erosion risk and information on short- and long-term erosion control programmes;
–the Ministry of Agriculture and Forestry is obliged to prescribe measures for preserving the soil from erosion by water and wind and to apply obligatory limitations on the landuse when land deterioration is established.

The Regulations for the enforcement of the Law on Protection of Agricultural Land (Anon., 1996b) set up the terms for obligatory landuse limitations on lands with high erosion risk and identifies the approach for establishing the soil erosion risk.

The Environmental Protection Act, which was approved on 1991 (Anon., 1991b) and expanded and changed 17 times up to 2001, regulates matters such as (i) collecting and submitting information on the environmental status; (ii) estimations of the environmental impacts; (iii) design and implementation of measures for environmental protection; (iv) rights and duties of the State, the municipalities and the physical and the juridical persons concerning environmental protection issues.

The major legislative measures to combat erosion in forest land have been laid down in documents regulating the requirements for planning, implementation, approval and maintenance of erosion control projects in the forest as follows:

- The Forest Act (Anon., 1997b) and the Regulations for its enforcement (1998b);
- The Law on the Reinstatement of the Ownership on Forestlands (Anon., 1997a) and the Regulations for its enforcement (1998a);
- The Instructions on Combating Soil Erosion in the forestlands (Anon., 2000).

The activities relevant to research, survey, construction and maintenance of systems and facilities to control landslide, abrasion and riverbank erosion are regulated by the Physical Planning of Territory Act (Anon., 2001a), Regulation No. 1 on geological protection activities (Anon., 1994) and Regulation No. 12 on design of geological control constructions, buildings and facilities in regions with landslides (Anon., 2001b).

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1.15

Moldavia

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1.15.1 INTRODUCTION

For centuries, humans have been attracted to this unique country by its natural beauty, fertile soils, temperate climate, green meadows, woodlands and water resources. The land is densely populated (4.4 million people) and has been economically utilized for a long time and therefore has been subject to significant changes. Natural conditions, considerable anthropogenic stress on the topsoil, intensive land use and almost 700 years of farming have resulted in land degradation.

One of the main tasks for erosion specialists, farmers and land surveyors is to find new methods for organizing farmland in accordance with conditions on the slopes. These issues are more important today because of processes taking place: changes of ownership relationships in rural areas, development of new management practices and formation of farmer and joint-stock enterprises. Because of the switching to the new economic approaches in farming, agricultural specialists and scientists face new challenges in erosion research and it is evident that assessing and preserving the natural resources and improving soil and water conservation practices are required. Therefore, measures for rational land use to increase soil fertility are of top priority and represent a major part of national policy.

1.15.2 EROSION PROCESSES: STATE OF KNOWLEDGE

The Republic of Moldavia is located mainly between the Prut and Nistru rivers. Lack of variety in its natural conditions and resources results from its small size (33 700 km^2). However, the hilly relief, heavy rainfall, low erosion resistance of the soil and unwise farming over many decades have caused high soil erosion rates, severe gullying and active landslide processes.

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Soil erosion was mentioned for the first time in papers by Grossul-Tolstoy (1868), Shmidt (1868), Pomompsestov (1868), Masalskiji (1897), Porucic (1916) and others. Gullies were considered as a 'terrible evil' in some areas. Dokuchaev (1887) mentioned the character and scale of soil erosion in Basarabia (the former name of the Republic). After World War II, Dimo undertook research on erosion processes. As a consequence, in the 1950s the Moldavia Branch of the USSR Academy of Sciences set up a station for soil erosion control, which was later transformed into the department of soil protection. Konstantinov (1958) released some maps of gully density and Gorbunov (1961) collected data on the gully networks within southern counties of the Republic and Transnistria. Systematic studies of gullies and the development of effective control methods were started in 1965. The number of gullies, their total length, morphometric parameters and the rate of gully growth were analysed and the technique for shaping of gully banks was developed by Yakovlev (1979).

Moldavia was a leading republic in the former USSR in terms of research on soil erosion and its control. Since 1970, Moldavia has been an experimental base for developing technologies for intensive crop production. However, misunderstanding of the natural conditions such as local hilly relief and heavy rainfall has resulted in degradation of land resources and a decline in land fertility. 'One-size-fits-all' application of intensive technologies to growing crops on the hillsides, expansion of the arable land to the detriment of forest, ploughing steep slopes and the underestimation of land improvements led to increasing erosion processes and reviews of conservation practices developed earlier.

Our research was carried out in two main directions: first, the identification of regional features in the development of erosion processes and how these features influence topsoil on the slopes subjected to gully erosion; and second, the improvement and development of new methods for recovering the fertility of eroded land.

1.15.3 EROSION FACTORS

1.15.3.1 Geology

A striking resemblance of the geological conditions with those from the Moldavian Plateau of Eastern Romania is obvious. The Moldavian Plateau lies astride the Prut River and it is mostly underlain by the border of the East European Platform. In Moldavia, the Precambrian basement is cut by the Nistru Valley and dips to SSW to a depth of over 2000 m. Most of the country is covered by clayey to sandy Miocene and Pliocene sediments with a gentle dip to the SSE. Shallow inter-beds of Sarmatian (Upper Miocene) sandstone and limestone can be identified (Ungureanu, 1992). Also, a Quaternary mantle of loess and loess-like deposits is noticeable. This mantle is shallow and discontinuous in northern and central Moldavia and more consistent in the southern area. Other conditions affecting the development of gullies, landslides and river erosion are tectonic fractures and local anticlinal structures (Voloschuk, 1978).

1.15.3.2 Geomorphology

Most of the area is included in the Moldavian Plateau (Ungureanu, 1992). The most typical subunits in central Moldavia are the Central Moldavian Plateau with a highest altitude of 429 m, the Nistru Plateau and the Ciuluc-Solonet Plateau (Figure 1.15.1).

The main relief features consist of structural platforms with the more resistant units capping hill tops. The second plateau unit is the Podolic Plateau that covers a narrow strip eastwards of the Nistru River with a highest altitude of 275 m. The southern and south-eastern parts of Moldavia are plains. First, the Bugeac

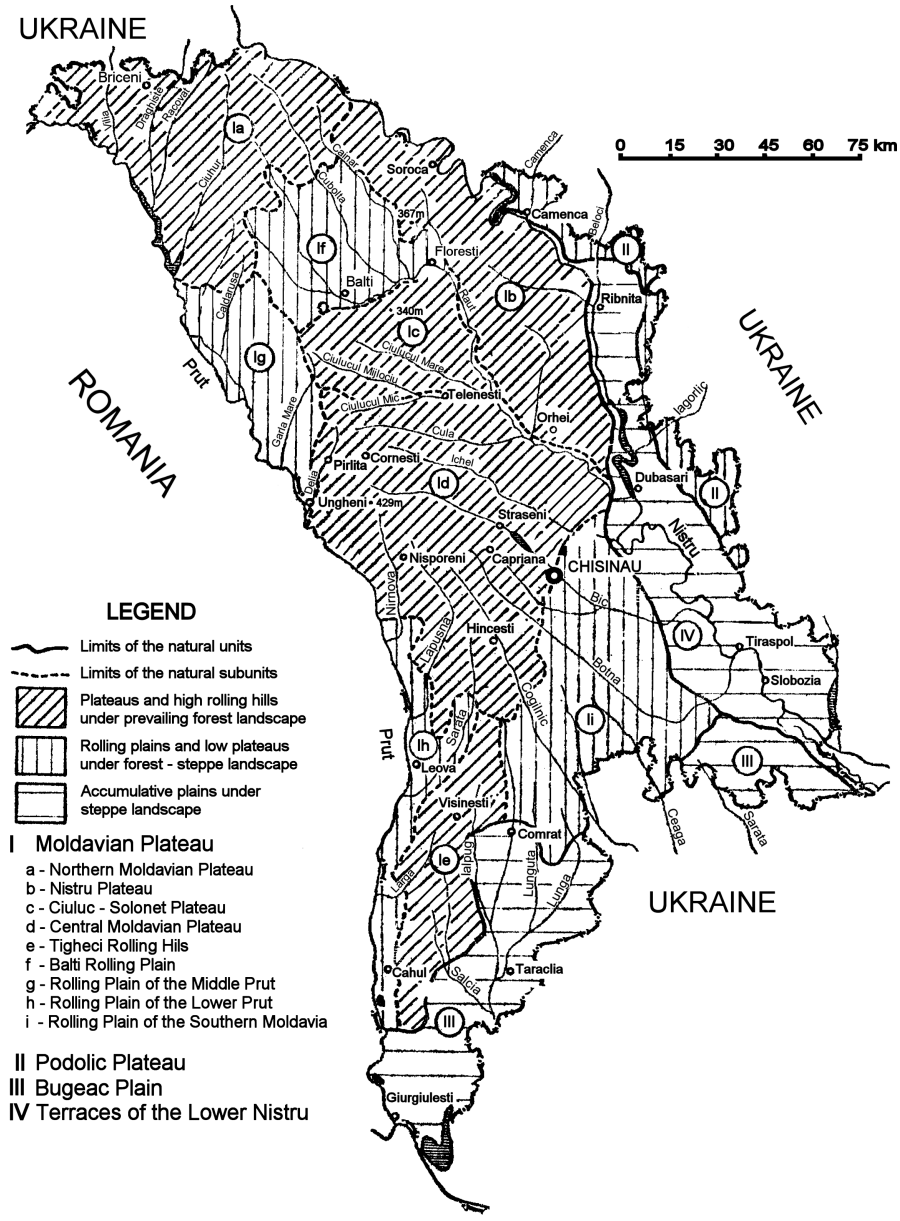

Figure 1.15.1 The physical–geographic units of the Republic of Moldavia. (After Ungureanu, 1992. Reproduced by permission of Dr Al Ungureanu)

(Bugeac means pasture field or steppe) is a typical fluvio-marine accumulative plain covered by a thick loess mantle where the altitude does not exceed 200 m. Second, the plain of the lower Nistru has an outstanding development of 11 terraces.

Catchments were classified on the basis of their genetic characteristics and morphometric parameters into 10 types, each with their own lithology and morphology. The density of the valley and gully network

plays an important role in erosional processes. There is a close relationship between soil loss and the density of valleys and gullies. In the south-eastern part of the Nistru valley there is no soil erosion on slopes or gullying. The Central Moldavian Hills are the most dissected territory. There one can clearly see a strip that stretches from north-west to south-east (from Falesti to Hincesti) and an average density of valley and gullies of 3–4 km km^{-2} reaching a maximum of around 5 km km^{-2}. In the Podolic Plateau (Transnistrian Hills) the value is 2–4 km km^{-2} (Figure 1.15.2).

Of interest for erosion processes is the depth of the local base level of erosion and its relationship to the intensity of soil loss and gullying (Sobolev, 1948). However, our research has shown that this relation is not universal. The closest relationship was found in the Central Moldavian Plateau, Transnistria Hills and Tigheci Rolling Hills, where the relief energy is around 200 m and the value of gully density is high. In the Northern Moldavian Plateau, Balti rolling plain, middle Prut plain and the plain of lower Nistru, there is low relief energy and very small areas of eroded land.

Many researchers agree that the most important relief features affecting surface runoff are slope length, slope steepness and slope shape (Zaslavskiy, 1979, Rojkov, 1981).

Research was carried out in nine key river basins in different areas (Dragiste, Ciulucul Mare, Bucovat, Isnovat, Rezina, Ribnita, Comarovca, Larga and Cahul) to establish the correlation between slope length and the area of eroded land and the number and total length of the gullies. Each parameter shows the specific features of each basin. For example, within the Rezina, Isnovat and Larga basins, the gully ratio increases with increase in slope length up to 750 m, whereas in the Bucovat and Ribnita basins this trend is up to 1000 m.

The average number of gullies is 1.63 km km^{-2}, varying between 1.23 and 1.86 km km^{-2}. In other cases, the number of gullies decreases with increase in slope length. In the Cahul, Dragiste and Comarovca basins, about 27–50 % of the gullies are developed on slopes longer than 100 m but the gullied area covers only 10–15 % of the catchment. Moreover, in the Ciulucul Mare basin, there is no direct correlation between slope length and gully density. Except for the Cahul basin, slopes shorter than 250 m and longer than 1500 m occupy less than 10 % of the total area and gullies are very rare.

Data on slope length influence on gully development have shown that slope angle has a direct impact on soil loss. Many researchers have considered the influence of slope shape on gullying (Sobolev, 1948; Kozmenko, 1957). Most of them concluded that on the longer south-facing slopes the erosional processes are more active than on the shorter north-facing slopes. However, these statements are not substantiated by data on gully density.

According to Voloschuk (1978, 1986), the peak gully density for most catchments occurs on the north, north-east, east, south-east and north-west-facing slopes. The south and south-west-facing slopes have the least number of gullies. Asymmetry of river valleys is very important for the development of gullies on north-facing slopes.

Ionita (2000a) emphasized a double system of cuestas that reflects two basic structural patterns in the Moldavian Plateau. The first type of structural asymmetry, associated with dipping of layers to the south, is responsible for the development of the classical north-facing cuestas along the subsequent valleys. The second type of structural asymmetry consists of gentler slopes on west-facing cuestas along most consequent valleys in the southern part of the Moldavian Plateau. In the central and northern Moldavian Plateau with more developed river systems, this particular feature was also found on reconsequent tributaries (Figure 1.15.3).

The western facing cuestas are caused by tipping of the Moldavian Plateau resulting from the impact of the Carpathian Mountain uplift during late Pliocene and Pleistocene. The most severe land degradation in this broader area occurs on north- and west-facing slopes. However, gully development was strongly influenced by the road network, usually up-and-down hill or along the valley bottom (Ionita, 2000b).

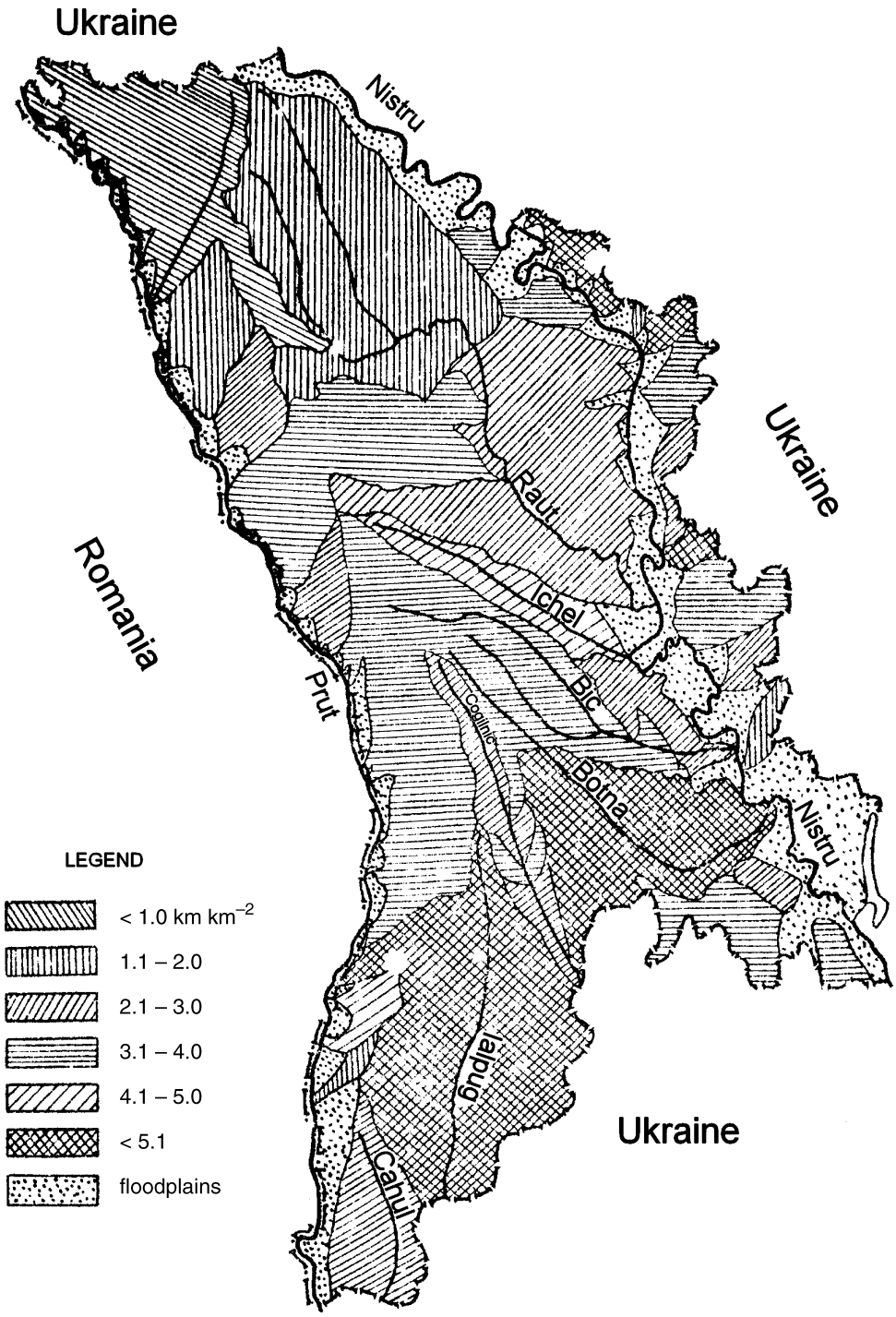

Figure 1.15.2 Density of valleys and gullies in the Republic of Moldavia (After Voloschuk, 1978, reproduced with permission from Cartea Moldovei)

Figure 1.15.3 The upper Lapusna basin, 10 July, 1995. (Photo I Ionita)

1.15.3.3 Climate

The climate is temperate continental with a mean annual temperature of 7.5 °C at Briceni in the north and 10.5 °C at Giurgiulesti in the south. The average annual precipitation varies between 370 mm in the southern plains and 550 mm in the higher plateau area. Of particular interest is the amount of precipitation per event and its frequency. Sometimes the amount of precipitation can reach 100–180 mm during a heavy rainfall. Two or three such intensive rain storms cause 80–90% of the annual soil loss. Gullying may be initiated on arable slopes in late winter with significant snow cover and quick snowmelt.

Based on long-term monitoring (1981–96) of continuous gullies in the Moldavian Plateau of Eastern Romania, Ionita (2000b) has emphasised that 57% of the total gullying occurred during the cold season, mainly due to freeze-thaw cycles. This ratio was higher in the case of discontinuous gullies.

1.15.3.4 Vegetation and Soils

There are three major areas, namely the steppe, the forest–steppe and the forest. The natural forest cover was severely modified by human activity and at present woodland averages only 9% of the total area. The soils are the most important natural resource of Moldavia and have been carefully studied. Their distribution is closely associated with the relief form, climate and vegetation type. Soils of grassland, the Mollisols, are most extensive, covering 75% of Moldavia, followed by brown and grey forest soils (12%) and alluvial soils (Krupenikov and Ursu, 1985).

Given the similar slope steepness (5–9%) and agricultural use, Mollisols are usually eroded modestly compared to grey forest soils. If one compares maps of distributions of gullies with soil maps, one can see a direct connection between the two. Among the most affected by gullying are slopes with loamy and sandy loamy soils. Gullies are mostly concentrated in areas with little or no forest (Voloschuk, 1978). Maximum annual gully head retreat varies from 0.76 to 3.5 m in areas under sandy loams and sands with a peak of 14.6 m. On loamy and slightly loamy sediments this rate was frequently between 0.66 and 1.8 m with lower

values on clayey and heavily loamy soils (Voloschuk, 1986). These values are in agreement with those presented by Ionita (2000b) for the Moldavian Plateau of Eastern Romania during the 1980s.

A study was carried out to assess the influence of windbreaks planted near gullies on gully growth in southern Moldavia (Voloschuk and Rojkov, 1970). This demonstrated that 60% of gullies under forest had ceased to grow, 23% had slow growth and only 18% showed high average annual growth. A similar influence of afforestation on the decrease in gully growth occurred in central and northern Moldavia (Fedotov, 1980).

High erosion rates may occur in areas where natural vegetation is thin or absent. For example, the gully systems developed on a pasture near the village of Cikur-Mingir, Cimislia county, increased from 1.5 ha in 1950 to 15.0 ha in 1980. On the other hand, 76% of the gullies are developed on pastures around settlements, 4% are on arable land and 20% are influenced by forest plantation (Voluschuk and Osadchaya, 1983; Osadchaya, 1985; Voloschuk, 1986).

1.15.3.5 Eonomic Activity

At present, soil erosion in Moldavia is mostly connected with improper farming on slopes. These processes are obviously on slopes steeper than 5%, where intensive farm practices are used. The soil erosion rates here have increased 8–10-fold compared with plots where corn is grown using traditional tillage, and 4–5-fold in vineyards where herbicides are applied instead of cultivating between rows (Fedotov *et al*., 1985).

The faster development of erosion in the south-western part of the country is due to the unfavourable combination of the natural and anthropogenic conditions including higher rainfall aggressiveness, hilly topography, a high ratio of arable land (over 80%), improper farming practices and low erosion resistance of the bedrock (sandy layers).

1.15.4 EROSION PROCESSES

1.15.4.1 Soil Erosion

The total area of the land subjected to soil erosion is 876 000 ha or 38.5% of Moldavia. Slightly eroded soils occupy 465 400 ha and medium and severely eroded soils 410 500 ha. Most of the latter are on slopes of southern and central Moldavia (Figure 1.15.4).

Research on eroded soils and their connection with the slope morphometry (slope steepness, length, shape and soils) was carried out in seven representative basins and a map of the erosion rates for each basin was produced. The maps show a strong relationship between the slope steepness and area of eroded soils. For example, the correlation index is 0.90 in the Motsa basin in Central Moldavia. Similar values were obtained for other basins (Voloschuk and Makhlin, 1974). That relationship between the area under soil erosion and slope steepness can be expressed by the equation

$$P = a - bI^2 + c$$

where P = the area of the eroded soils (%), I = slope steepness (degrees) and a, b, c = regression coefficients.

A clear dependence between slope length and area with soil erosion has not been revealed in all natural regions. However, the Motsa, Dragiste, Salcia Mare and other basins show a trend with an increase in the area of eroded soils with respect to slope length up to 800 m. Then follows a gradual decrease that apparently results from a decrease in slope steepness and reduced runoff. The value of the correlation ratio is 0.76.

The relationship between slope length and area subjected to soil erosion can be expressed by the equation

$$S = aL^h$$

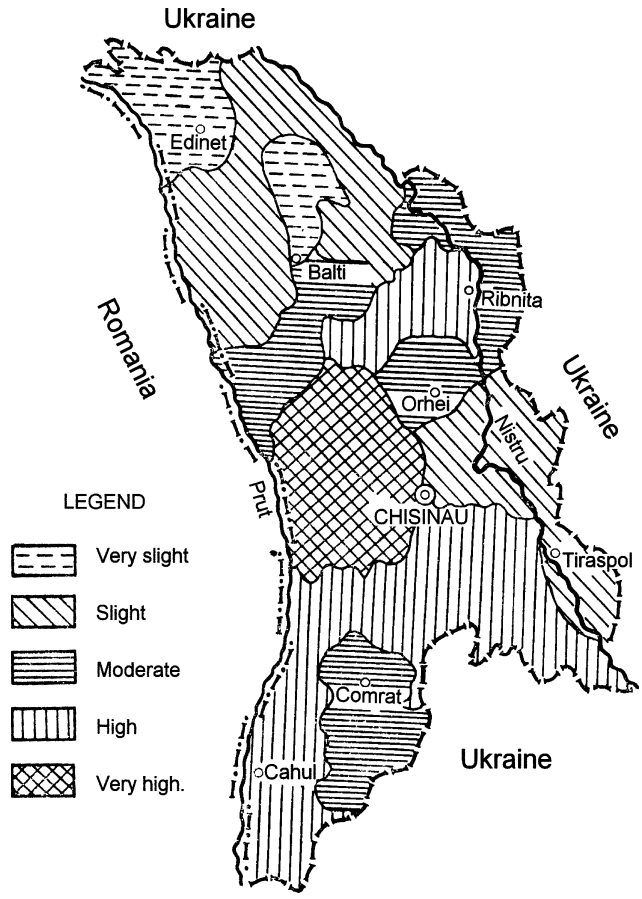

Figure 1.15.4 Soil erosion risk on agricultural land in the Republic of Moldavia. (After Konstantinov, 1987. Reproduced by permission of IS Konstantinov)

where $S =$ the area of eroded soils (%), $L =$ the slope length (m), $a =$ dimensionless coefficient and h reflects the runoff velocity, which is dependent on increasing slope length. For example, $a = 9.1$ and $h = 0.31$ for the Motsa catchment.

The general relationship between the geomorphological features and the area (number) of eroded soils in the basins can be expressed by the following equations:

$$y = ax + b$$
$$y = ax^2 + bx + d$$
$$y = ax^b$$
$$y = c(1 - e^{-bx})$$

where $y =$ the area of eroded soils, $x =$ the corresponding morphometric parameter and a, b and d are empirical non-dimensional parameters that characterize the physical–geographical conditions of a specific basin and are determined using the least-squares method.

If the mean-square error is taken as a measure of the model accuracy, then for the Motsa basin the last model fits best, regardless of what was chosen as a morphometric parameter of the relief (either slope length or steepness). The influence of slope length on the area prone to soil erosion in the Salcia Mare basin is expressed by the first model, that is, by a simple linear dependence. According to the most reliable models, the calculations show that there is good agreement between observed (measured) and estimated data (Voloschuk and Makhlin, 1974).

The correlation between the slope aspect and the area of eroded soils was evaluated by the value of the mutual contingency coefficient. Areas with eroded soils mainly correspond to west-, north-west-, north- and north-east-facing slopes (north- and west-facing cuestas) and much less to south-west-, south- or south-east-facing slopes.

1.15.4.2 Gully Erosion

The gullies are most widespread and most actively developing in southern Moldavia and in the Camenca area in northern Transnistria. The density of gullies varies from 0.7 to 1.0 km km^{-2}. The most severe gully erosion occurs in the basins of the Lunguta, Lunga, Salcia Mare, Salcia Mica, Ialpug, Larga, Ribnita and Camenca rivers, followed by central Moldavia, Tigheci Rolling hills and Transnistria (Podolic Plateau), where the gully density is 0.2–0.5 km km^{-2}. In this area, most of the gullies are located in the upper basin of the Bic, Bucovat, Nirnova, Lapusna, Ciulucuri, Cogilnic and Isnovat rivers. The smallest gully density of less than 0.23 km km^{-2} is typical of northern Moldavia, but there some spots show much higher values.

The issue of classification of the relief forms associated with erosion was considered by Masalskiji (1897), Kizenkov (1902), Kern (1928), Zanin (1952), Braude (1959), Armand (1972), Shvebs (1974) and Lidov (1981). The basis of our classification is lithology, the stages of gully development and the morphometry of the gullies and basins (Voloschuk and Djemelinski, 1975). According to age and morphometry, gullies can be divided into pre-anthropogenic and anthropogenic. The latter are further split into three subgroups according to their stage of development, namely initial, transitional and mature. Depending on location, the last subgroup consists of different types, subtypes and varieties.

Contemporary gullies can be separated into two types according to their position within the catchment. The first type includes those gullies located on valley sides. The second type comprises valley-bottom gullies that are characterized by significant length, drainage area and low longitudinal gradient.

Data from 15 000 gullies have been collected for studying gully morphometry (length, width, depth, area, volume and catchment area). For each parameter the average values, the mean-square deviation and different coefficients such as variation, excess or asymmetry were calculated. Also, they were grouped and distribution curves were plotted (Voloschuk, 1986).

The quantitative characteristics of gullying and distribution patterns substantially saved time and money for field work when developing the ‘General draft for erosion control over the period 1991–2005’. Moreover, these data were useful for choosing more confidently the best practices, to assess the amount of work and the required cost of gully erosion control.

In order to monitor gully growth, a system of 256 fixed points (96 on hillsides, 143 on watersheds and 17 in valley-bottoms) was set up in 1966. Observations on gully width and the volume of soil loss were made in five representative gullies. Average annual gully-head retreat varies between 0.66 to 1.24 m with a peak of 7.51 m for valley-side gullies over the last 25 years. The gully retreat usually occurs during spring and summer. When there was a lot of snow, as in 1973, 1977 and 1985, the rate of gully growth was 2–3 times higher during autumn and winter than spring and summer. On average, the valley-bottom gullies develop with approximately the same intensity from both the snowmelt and rainfall runoff. Their seasonal growth is 2–3 times higher with respect to the valley-side gullies. The average annual retreat of valley-bottom gullies was 4.0 m. In terms of how active the gully growth of the 256 gullies is, about 30% are slowly, 25% moderately and 45% very

TABLE 1.15.1 Gully development in the agricultural land of Moldavia

Indicator	1911	1965	1982
Total number	10675	41517	6205
Total area (ha)	16314	21473	5785
Total length (km)	—	13057	2035

actively growing. There is a direct relationship between gully growth, slope steepness, catchment area and the amount of precipitation. The coefficients of the multiple correlation are 0.87, 0.90 and 0.91.

Large-scale maps and air photographs were used to study gully growth of some gully systems within the Southern Moldavian rolling plain, the Central Moldavian Plateau, Tigheci rolling hills and Podolic Plateau (Transnistrian hills). Three major inventories of the gullies in Basarabia (the former name of the Republic of Moldavia) were undertaken as follows: the first in 1911–15, the second in 1965 and the third in 1982. These surveys enable us to evaluate the spatial distribution of the gullies and changes in the gully landscape over the last 70–75 years with the compilation of maps of gully density (Voloschuk and Zagarovsky, 1981).

According to these successive gully surveys, the number of gullies increased 3.5-fold and their area 1.5-fold between 1911 and 1965. That means that a lot of small gullies appeared (Table 1.15.1). There was a drastic decline in the number of gullies and the area they cover from 1965 to 1982 due to significant amounts of land reclamation work. However, despite this fact, most of the area between the Prut and Nistru is still prone to active gully advance of $1\ m\ yr^{-1}$ on average. Generally, every year around 700–800 scours (head-cuts), ephemeral gullies and gullies are formed with a total length of 60–70 km covering about 300 ha and damaging 450–500 ha of land (Figure 1.15.5).

Some data on the need for assessing and mapping gullies were published by Kozmenko (1957), Jilko and Lemeshev (1972), Rojkov (1973) and Zaslavskiy (1979). Different gully forms were identified by Voloschuk and Petrov (1981) and their morphometric parameters were mapped (Voloschuk and Zagarovsky, 1986). Within the Isnovat basin, from the Central Moldavian Plateau, of 256 studied hillside gullies and their side-gullies, 40% are situated closer than 100 m to each other, 30% are separated by 100–250 m, 20% by 250–500 m, 8% by 500–1000 m and 2% are over 1000 m apart. In the Valea Rezina catchment in Transnistria, of 102 hillside gullies 50% are situated closer than 100 m to each other, 10% are 100–250 m, 6% are 250–500 m, 30% are 500–1000 m and 14% are over 1000 m apart. The inter-gully area is around 1 ha in 45%

Figure 1.15.5 Active gully-head

of the cases, it accounts for 2–3 ha in 30% of cases and it is over 3 ha in 25% of the cases. Gully density is closely connected to human activity, to the presence of natural and artificial boundaries that cause increased concentrations of the water flows and to high rates of erosion.

1.15.5 RECOVERING THE SOIL FERTILITY ON SLOPES

Since 1970, a theoretical basis and various technologies for soil recovery have been developed based on the use of modern reclamation machines rather than manual labour. A major reconstruction was needed for recovering the productivity of slopes subjected to soil erosion by replacement of humus layers (Krupenikov and Lejb, 1965).

The basic materials for recovering fertility of low-productive lands are the recent sediments deposited on the floodplains and the impressive network of reservoirs. Also, a lot of soil can be collected from the processing plants in sugar beet areas. The most widespread reserve is deposited soil, which amounts to 94 400 ha, and Transnistria (37 200 ha) and Central Moldavia (20 400 ha) should be mentioned first. There are over 3500 reservoirs in Moldavia regulating over 800×10^3 m^3 of surface waters. About 60% of them are silted up and their annual loss of volume is 4–5%. The 36 large reservoirs in the southern steppe and central forest–steppe contain around 930 000 t of valuable organic matter, nitrogen, phosphorus and potassium. These materials can be used either separately or in different combinations with additional organic fertilizers.

The main criterion for assessing the need for recovering soil fertility is the thickness of the soil profile. There are five land types that should to be treated:

- the eroded soils and gullies that cover an area over 350 000 ha and represent the main priority for land improvements;
- slopes dissected by gullies that are needed for agriculture;
- agricultural areas under landslides;
- land disturbed by industrial activities;
- areas with outcropping bedrock.

Three field experiments were carried out to develop technologies for improving soil fertility using the replanting (transplanting or coating) method. They were also used to determine the effectiveness of various kinds of humus layers and their influence on improvement of soil qualities. The working hypothesis was to determine the optimum thickness of the applied humus layers (15, 30 or 45 cm) and the degree of their similarity to the initial full-profile layers in terms of fertility.

The technical elements of the soil recovery include control of surface runoff on the slopes by applying simple conservation measures, levelling of the slope, deep ploughing before replanting with new material, selective application of the humus layers accompanied by ploughing and putting the field under perennial grass for 2–3 years.

We have developed and tested technologies for improving soil fertility and for soil erosion control for each of the slope categories affected by gullying. For the roughness categories I–III a system of conservation measures would prevent initial linear erosion and improve fertility. Experiments were carried out in the Transnistria on a slope 700 m long, east-facing and with a maximum steepness of 14.5%. Every 20–60 m it was cut by gullies of average depth 0.8 m, width 12 m and length 300 m. The soils between the gullies spaces are carbonate-rich loamy Mollisols, of which over 60% are moderately and heavily eroded. At the bottom of the slope and along the gully bottom there are newly deposited soils with average layer thickness (A + B horizons) up to 210 cm. The survey was carried out according a plan of full backfilling and flattening of the gullies by the deposited soils found at the base of the slope. The total area of the plot was 17 ha.

Observation of the effectiveness of the methods showed that after 18 years there was no erosion due to heavy rainfall. The crop yield data emphasized that the most effective practice was the full backfilling of gullies with soil taken from the slope base. The increases in yield were sunflowers 4.1, winter barley 5.8, winter wheat 5.2 and maize green mass 68 kg ha^{-1}. For category IV soils more affected by deeper gullies (1.5–2 m) on slopes of 9–13%, a technology of slope levelling and conserving the fertile layer of soil was tested. This included the following operations: construction of water-regulating earthworks, separation of the slopes into working plots, selective removal and storing the humus layers from the working plots, surface levelling, ploughing up and selective replanting (coating) with humus layers and establishment of ploughed-up embankments. These large-scale experiments were, however, accomplished in a period when fuel prices were very low.

The measures developed earlier to prevent gully formation proved to be of no use for solving the issues connected with land use of these areas. By the 1960s, the priority was to control actively growing gullies that threatened buildings, roads and other objects. Short and shallow gullies that accounted for about 60–80% of the total were out of control. Therefore, a need emerged to develop new technologies for controlling gullies.

Data on the gully backfilling were published by Masalskiji (1897), Rabcevich (1907), Zykov (1965), Odaciuk (1973), Rojkov (1973), Rojkov *et al*. (1982) and others.

New approaches were based on:

- storage of the topsoil in spoil banks and distribution of topsoil on the levelled area;
- newly cut slopes to remain under the angle of stability defined by the grain-size distribution of soil;
- creation of water-regulating earthworks (embankments) simultaneously with backfilling of gullies before topsoil is applied on the levelled surface.

For recovery of slopes, the following should be given highest priority:

- The gullies that are no more than 5 m deep and located on tilled slopes that are not steeper than 13%. If separate plots can be united into a single field, it is worthwhile to backfill gullies and create slopes of no more than 5.4% for annual crop rotation and no more than 13% for conservation crops.
- Slopes of up to 18% with gullies can be used for vineyards or orchards after full backfilling.
- Gullies with very steep banks developed in hard rocks (limestone, sandstone) which are designated for use as agricultural holdings or forest plantations should be filled or flattened by drilling and blasting operations followed by surface levelling.

We have developed this new technology for gully backfilling and implemented it into 55 projects. The usual steps involve stump and bush uprooting, stone removal, building earthworks to control peak flows, removal and storage of topsoil, reshaping slopes, drainage control in the gully bottom, building the water-regulating earthworks on the levelled surface, deep ploughing of the levelled plots, selective application of topsoil on the new surface and sowing perennial grass on new slopes and earthworks.

However, the present-day soil classifications do not account for anthropogenic soils that were formed by applying humus layers on eroded or levelled areas. The total area of such soils is about 60 000 ha in Moldavia.

1.15.6 ENVIRONMENTAL–ECONOMIC ASSESSMENT OF LAND TREATMENTS

The radical methods that were developed to recover fertility of the topsoil and to regulate surface and ground runoff are based on principles appropriate to the natural landscape and on an understanding of the long-term

environmental consequences of exploitation of low-productive lands. The engineering–economic basis for various methods for recovery of fertility on eroded soils is a complex application of these methods jointly with a system of soil conservation practices. Their economic effectiveness depends on many factors such as the amount of investment, the conservation techniques and structure of the agricultural holdings and the area of the land that is to be protected and the land that is tilled.

In order to prevent erosion and gullying in orchards and vineyards and to prevent deepening of gully bottoms, we suggest various types of constructions (dams). The pay-back period for their construction depends on the materials used and usually varies between 2 and 5 years.

When calculating the economic effectiveness of replanting (coating) eroded soils, there have been taken into account all the old costs based on the very low fuel prices. For specific areas affected by gullies, the investment pay-back period depends on the depth of the gullies and varies from 3 to 15 years. The pay-back period for gully reclamation work and for later land use as orchards and vineyards is 1–3 years after they start to bear fruit.

In Moldavia there are significant areas of low-productive fields that could be used as agricultural land. The fertility can be increased on 150 000 ha of moderately and heavily eroded soils if the topsoil is replaced by, e.g., deposited soils and sediments from floodplains and reservoirs.

1.15.7 CONCLUSIONS

The natural landscapes in Moldavia have been drastically changed by human activities during the last century, resulting in increased land degradation.

The highest risk of soil erosion on the agricultural land is associated with the Central Moldavian Plateau and resulted from the higher relief, large area of forest soils and the amount of precipitation.

There are over 55 000 gullies longer than 50 m and their total length is about 13 000 km. The average density of the gully network is 0.39 km km^{-2} or 1.63 gullies km^{-2}. The highest density and the most active gullies are located in southern Moldavia. Landslides mostly occur on cuestas facing north and west.

Best management practices have been deployed on a large scale. Of interest is a new technology for recovering soil fertility in areas subjected to gullying or landsliding consisting of backfilling of gullies, slope reshaping and levelling, deep ploughing before replanting or transplanting with new material, selective application of the humus layers accompanied by ploughing and putting the field under perennial grass for 2–3 years.

The main resources used for backfilling of slopes are alluvial soils, sediments deposited along the floodplains and reservoirs and soils from areas under civil construction. By combining this particular technology with classical agricultural conservation practices, a decrease of 80–85% of runoff and erosion is achieved.

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1.16

Ukraine

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1.16.1 PHYSICAL GEOGRAPHY

The area of the Ukraine is 603 700 km^2. Its territory spreads 1300 km eastwards (from longitude 22 to 40° E) and nearly 900 km southwards (from the latitude of 52 to 45° N). It is located in central and south-eastern Europe and borders Hungary, Slovakia and Poland in the west, Belarus in the north, Russia in the north and east and Romania and Moldova in the south. Its most southern part is washed by the Black and Azov Seas.

The Ukraine is mostly flat: nearly 90% of the whole area is plain; the average elevation of the flat area is 170 m. Mountainous areas occupy nearly 5% of the territory, namely the Carpathians (20 000 km^2, with some peaks 1700–2000 m above sea level) in the west and the Crimean Mountains in the south (5 000 km^2, reaching over 1500 m above sea level).

The climate of most the Ukraine is continental, varying from low continental in the west and northwest to medium continental in the east and southeast. Only a narrow strip in the southwest of the Crimean Peninsula is characterized by a subtropical climate. Annual precipitation on the flat part of the territory is from 300 to 350 mm in the south, from 700 to 750 mm in the northwest, over 1200 mm in the Carpathian Mountains and from 800 to 1000 mm in the Crimea.

Erosionally hazardous climatic events sometimes occur. Thus, since 1971 maximum rainfall events were observed at weather stations in Poltava (rainfall intensity 4.65 mm min^{-1}, rainfall duration 2 min), Lubny (4.2 mm min^{-1}, 2 min), Kamenets-Podolskiy (3.17 mm min^{-1}, 3 min), etc. The seasonal distribution of erosionally dangerous events is as follows: 58% (>10 mm) and 66% (>20 mm) during the summer season, 23% (>10 mm) and 16% (>20 mm) in spring, 19% (>10 mm) and 18% (>20 mm) in autumn. From spring to autumn in the southeast and southern parts of the country there are sometimes droughts, dry winds (25–30 days per year), and dust storms (3–8 days per year). The duration of dust storms varies from a few minutes to a few days. In the north (Polesie area), rainfall and snowmelt soil erosion are comparable. In the forest–steppe area

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the ratio between the rainfall and snowmelt soil erosion is 2:1. In the south, the contribution of snowmelt erosion to annual soil erosion loss is less: in the arid steppe, snowmelt erosion is negligible. Overall, the erosive effect of rainfall is 10 times greater than that of snowmelt.

1.16.2 SOIL

The topsoil of the Ukraine is very diverse. According to the national soil classification, it includes 650 types and the total number of soil varieties is several thousand. The distribution of soil types over the territory is closely related to the other elements of the environment, such as physical geography of the location, climate and vegetation.

The agro-pedological zoning of the Ukraine reflects the most general features of the main agro-pedological groups (Table 1.16.1). Names of soil types are given according to the national system (Polupan, 1988).

Polesie, which occupies 14.5% of the total area, is the area of Sod-Podzol soils lying on flat, runoff affected land. In the forest–steppe, the topsoil structure is determined by the diversity of plants, of climate and geomorphological conditions, and also by differences in land use and land management practices. In the steppe area, the heterogeneity of topsoil is basically determined by climate conditions and plant cover.

In the Crimean Mountains, vertical zoning and, connected with it, heterogeneity of soil-forming factors determine the noticeable complexity of soil cover. The basic soil in the steppe foothills is Chernozem lying on the regolith of various dense carbonate parent materials and clays. The foothill forest–steppe is represented by Sod carbonate and grey forest steppe soils. In the forest mountain area, brown soils are developed and on the tablelands meadow Chernozem-like soils occur. The subtropical southern area is occupied by brown soils.

1.16.3 LAND USE

Agricultural lands occupy 70.3% of the total area of the country and cultivated lands 81% of the agricultural area. The most widespread soils under agricultural land use are Chernozem (60.6%) and dark grey forest soil (21.3%). The steepness of agricultural land is differentiated as follows: from 0 to 1.3°, 78%; from 1.3 to 3°, 17%; from 3 to 6°, 0.9%; from 6 to 12°, 2.1%; from 12 to 20°, 1.8%; and above 20°, 0.2%. Varieties of the

TABLE 1.16.1 The main characteristics of soils in the Ukraine (Polupan, 1988)

Soil type	pH	Cation exchange (Mgeq per 100 g)	Hydrolytic acidity (Mgeq per 100 g)	Base-saturated degree (%)	Humus (%)	Bulk density of soil (g cm^{-3})
Turf–Podzol soil	6.3	4.4	2.3	75	1.3	1.5
Turf soil	5.7	3.3	2.0	62	2.2	1.5
Light grey soil	5.1	16.5	3.2	81	4.2	1.35
Grey soil	4.5	20.0	3.5	85	2.0	1.4
Dark grey soil	7.0	31.5	3.9	89	7.3	1.1
Podzol Chernozem	7.0	22.3	3.3	88	5.2	1.2
Typical Chernozem	7.0	36.3	0.7	98	5.5	1.2
Common Chernozem	7.2	37.1	1.1	97	5.0	1.2
South Chernozem	6.9	36.4	1.6	96	3.6	1.4
Chestnut soil	7.4	26.5	1.8	95	3.4	1.2

natural and economic conditions determine different land-use practices and regional specialization in agriculture. Polesie, the most part of which is covered by forests and shrubs, is characterized by large areas of rangelands, while the portion of cultivated lands is relatively small. In the forest–steppe, where natural conditions are very favourable for agriculture, the area of cultivated land and hayfields is very high. In the steppe area, which occupies 41.5% of the whole territory, agricultural lands occupy over 76% and cultivated lands occupy near 63%.

The Ukraine accounted for 2.7% of the land of the former USSR and had 15.1% of the total arable land with 25% of the total agricultural production.

1.16.4 HISTORICAL EROSION

There have been three stages of loss of organic carbon from soils in the Ukraine:

1. 10 000 years – 31×10^6 t yr^{-1};
2. during the last 300 years – 300×10^6 t yr^{-1};
3. During the last 50 years – 760×10^6 t yr^{-1} (24 times the historical average)

Erosion is a disaster that did not arise in the Ukraine by accident, but is the natural and inevitable result of mismanagement of agricultural production systems.

1.16.5 CURRENT EROSION

According to data from the Ministry of Agriculture, about 500×10^6 t of soil on average are lost from the Ukrainian arable land yearly (Figure 1.16.1). Simultaneously, 23.9×10^6 t of humus, 964 000 t of nitrogen, 676 000 t of phosphorus and 9.7×10^6 t of potassium are lost. The yearly soil loss averages 7.7–2.7 t ha^{-1} depending on region.

Erosion totalled 200 t ha^{-1} during one storm and even greater losses are common. The area of agricultural land in 1991 with some erosion was 12.1×10^6 ha (30.7% of total agricultural land); this included 9.4×10^6 ha of arable land. The area of eroded land increases at a rate of 80 000 ha yr^{-1}. Moreover, wind erosion processes also occur in the Ukraine. Some 19×10^6 ha (about 50%) of agricultural land in the Ukraine are subject to wind erosion, including 16.6×10^6 ha of arable land. About 5.9×10^6 ha of agricultural land are already eroded to a variable extent, including 5.4×10^6 ha of arable land. According to the data obtained from the Institute of Soil Conservation (Lugansk), the shortfall of grain production resulting from soil degradation is 8.6×10^6 t ha^{-1}, equivalent to a loss of US$20–30 million per year (Bulygin, 1994). There is evidence to suggest that the intensity of erosion is accelerating in spite of considerable attention of technical specialists and the public to the erosion problem (Bulygin and Nearing, 1999).

For each information unit which is an administrative region, average weighted values were estimated for runoff length, slope, soil erodibility, crop management factor – parameters involved in the hydromechanical soil erosion model (Mirtskhoulava, 1970). This model was modified and adapted to the task of the estimation of erosion process development (Bulygin, 1992; Bulygin *et al*., 2002).

According to the data, there is no soil erosion risk in the Polesie and dry steppe areas. On the map, the areas of the Carpathian and Crimean Mountains are also shown as areas with no soil erosion risk. This occurs because the mapping methodology is not appropriate. The greatest soil erosion risk in the Ukraine exists in the southern forest–steppe and northern steppe areas The data represented on the map are verified by the experimental data obtained from runoff plots during a number of studies by Ukrainian scientists.

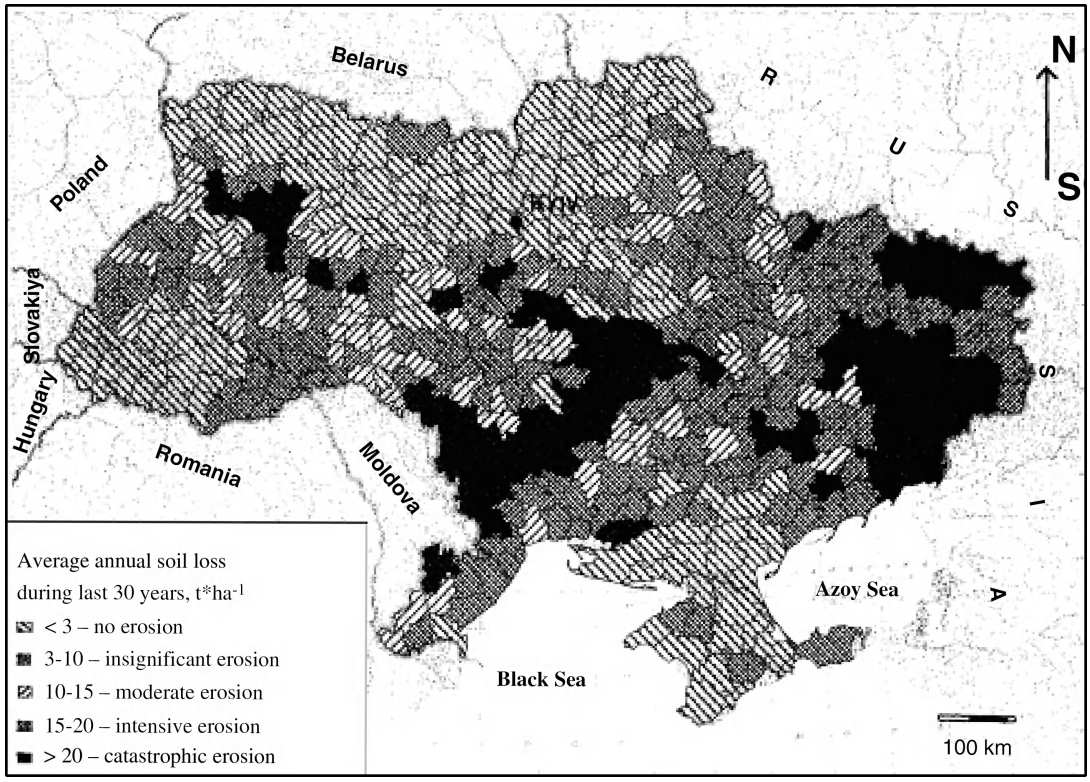

Figure 1.16.1 Loss of soil from Ukrainian arable land

A set of maps exist for the estimation of eroded lands; however, these maps cannot be considered seriously because they were built using an erroneous method (comparison of soils on slopes with the analogous soils on plains).

The causes of the accelerating erosion intensity are enumerated below. Economic resources allocated for soil erosion control have been used efficiently. The analysis of State investments in soil conservation has shown that during 1976–85 72.4% of total funds were spent on building anti-erosion hydrotechnical structures (so-called 'anti-erosion ponds', iron–concrete devices and so on), 10.5% on land restored wasteland, 5% on other works (including planning) and only 12.1% on erosion control (Jamal *et al.*, 1986).

1.16.6 IMPACTS AND COSTS

There are 24×10^6 t of humus, 1×10^6 t of nitrogen, 700 000 t of phosphorus and 10×10^6 t of potassium being lost yearly. There are on average 8–30 t ha^{-1} erosional losses from tilled areas yearly. We can often observe the catastrophic influence of erosion with the loss of 200 t ha^{-1} and more from fields. About 40% of agricultural land is at risk pf erosion. The total damage due to erosion is more then US$ 10 million per year, approximately equal to the national budget of the Ukraine. Nearly 19×10^6 ha of agricultural land is subject to deflation, including 16×10^6 ha which is tilled, with 5.9×10^6 ha of agricultural land already damaged including 5.4×10^6 ha of tilled land.

1.16.7 SOIL CONSERVATION MEASURES

There is an extremely high proportion of arable land in the territory of the Ukraine: arable land covers 56.9% of the whole territory and 81.1% of the agricultural land. This is the highest proportion of cultivated land in any European country, or in any developed country in the world. With such a high proportion of arable land, it is difficult to protect it from water and wind erosion processes.

Forestry as a method of protecting fields against erosion is undervalued. There has been a decrease in the use of forest strips and maintenance of existing ones. Protection strategies used on the plains have unfortunately also been used on slopes. The main mistakes are the use of rectangular-shaped fields and placement of protective forest strips and other stable borders along slopes, ignoring the laws of landscape development and transformation, in particular, ignoring the association of mass and energy exchange between landscape elements. In the Ukraine, with the exception of nature reserves, there are no natural landscapes.

Therefore, the problem will be solved only by constructing special anti-erosion agro-landscapes where water and wind erosion processes are common. The agro-landscapes should approximate the natural landscapes. With rare exceptions, the methods used to protect the landscape are not a part of a definite system and are without scientific substance. The use of full-scale soil-protecting technologies would lead to a sharp decline in soil loss caused by erosion.

A serious obstacle to the introduction of soil-protection technologies into the cropping system is the absence of soil conservation tools for cultivation. Techniques for managing crops on slopes of more than 3° (sowing, harvesting and plant care) are practically absent. Also, tractors are not adapted to work across inclined slopes. Therefore, according to the concept of contour-ameliorative agriculture of the Agriculture Institute (Kiev), row crops should be concentrated on land with slopes of less than 3° in the so-called first technological group, thus providing some protection against degradation (Tarariko *et al*., 1990). However, it seems that such a 'straightforward' recommendation cannot be applied everywhere. The absence of direct monetary interest of land users in soil conservation also affects their attitude towards the land. This is the basis of the contemporary harmful activity of humans which has a material and political basis. The absence of a land value appraisal has created a paradoxical situation – the means of production have practically no price. Land users have practically no responsibility for damage to the main production base – the soil. Their work is evaluated by the profitability of the production enterprise. Under such conditions, measures are carried out that increase crop yields in the year of their application. Soil protection is of low interest, as economic considerations are focused on immediate results. This is also a characteristic of Western countries with developed market economics.

Economic levers securing reliable and effective 'soil health' are necessary. Especially important is the development of accurate and clear techniques for the estimation of the losses caused by erosion. It follows that the development of an agro-landscape needs to include reliable protection from degradation processes such as erosion. This statement is strengthened by the obvious fact that halting erosion is a precondition for further improvement or restoration of soils, i.e. without solving the erosion problem, any plans for improving the fertility or other functions of the soil surface are doomed to failure. The development of an erosion-resistant and ameliorative agro-landscape at a particular site is an engineering process. Therefore, it must be based on some conceptual agro-landscape model, which must adequately reflect the peculiarities and intensity of the erosion process. An agro-landscape conceptual model is the general scheme of anti-erosion measures in a particular region. The development of a conceptual model seems to be quite conjectural without preliminary differentiation of the land by some soil erosion index. This has considerable difficulties: traditionally, it is carried out on the basis of maps on which actual soil erosion is indicated. We suggest using a potential erosion risk index. The map of the potential risk of erosion is radically different from the map of actual soil erosion. If the prevention of erosion is based on an actual soil erosion map, the main efforts will be in regions with highly eroded soil surfaces. In such an approach, the actual erosion frequently does not coincide with the potential

risk of the erosion process. As a result, in regions with moderate or low erosion, further intensification of agricultural use is planned, even though there is high potential danger of erosion. There is one more substantial detail that does not permit the management of agricultural lands only on the basis of maps of actual soil erosion. It concerns the difficulties in diagnosis of the erosion grade. This method is based on the choice of control being used in the watershed, for which the thickness of soil is compared with the thickness of soils on slopes. Such a method causes gross errors. For instance, weakly developed arable soils on slopes with southern aspects are diagnosed as eroded, whereas actually they are non-eroded.

1.16.8 LEGISLATIVE DEVELOPMENTS

For many years, problems of accelerated soil erosion were almost completely ignored by the government. Many attempts have been made to attract the attention of members of society, who are indifferent to the state of the Ukrainian lands and to the future of the Ukraine.

Eventually, the efforts succeeded, resulting in the development of the Land Code of the Ukraine, which was approved on 25 October 2001. The Land Code is a legal basis for land protection and land fertility restoration, which regulates all the main issues related to land conservation management.

Another important governmental document is the decree 'The main directions of the land reformation in the Ukraine in 2001–2005', issued by the President of the Ukraine. Along with many other points, this decree assumes improvement of the control methods on land use and land conservation, implementation of economic methods that will encourage land owners and land users in land conservation, etc. Also, increasing interest has been noted of farmers and land owners, who are seriously involved in agricultural production and who would like to protect and improve the state of their resources.

The next important step for the Ukraine towards land conservation development would be the creation of a Soil Conservation Service, which is currently in progress. This question has been raised by scientists at the National Scientific Centre 'Institute of Soil Science and Agrochemistry' and, seemingly, it has received a proper response from the Ukrainian government, which is now trying to find financial support to create a Soil Conservation Service. This would be a great advance for the Ukraine, which will allow development of a civilized land management system, improve the quality of life and preserve one of the most valuable Ukrainian resources for future generations.

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1.17

Austria

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1.17.1 INTRODUCTION

Owing to the special geomorphological situation of Austria, of which more than 60% is alpine territory with extremely high relief energies, erosion and erosion control have been a major issue for a long time. The focus of activities was and still is on torrent and avalanche control, as these are major threats to human life in alpine environments.

According to BMLFUW (2001), about 67% of the Austrian territory may be classified as either part of a torrent watershed, avalanche watershed or general risk area. A total annual budget of €70 million is planned to be invested in measures against these risks. However, although some reference will be made to erosion risk in alpine areas, the focus of this chapter will be on enhanced soil erosion due to human impact, that is, on-site soil erosion on arable land.

1.17.2 GENERAL ENVIRONMENTAL CONDITIONS

Owing to the different landscapes of Austria, all factors that may influence soil erosion exhibit enormous temporal and spatial variations.

Long-term annual rainfall (1961–90) varies between 430 and 2250 mm, with an overall mean of about 1170 mm. This corresponds to theoretical *R*-factors (according to the USLE) of 38 and 180 $N\,h^{-1}$ (Strauss *et al.*, 1995). However, intensive agricultural land use to which the USLE calculations are limited is not practised above about 1500 mm of annual rainfall. Especially in the transition zones between alpine and lowland areas, spatial climatic variations may be remarkable. Within a distance of 50 km rainfall varies from

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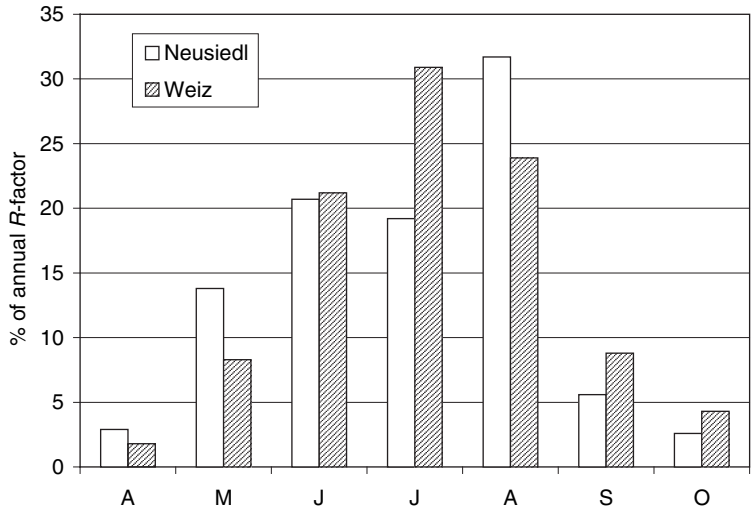

Figure 1.17.1 Seasonal distribution of *R*-factors for two Austrian stations. (After Strauss *et al.*, 1995)

720 mm at Petzenkirchen to about 1780 mm at Neuhaus (BMLF, 1994). Temporal changes of driving forces for erosion also exhibit distinct variations. Figures 1.17.1 and 1.17.2 give typical examples of temporal variations for erosive rainfall and wind speed.

The distribution of soil types reflects the different environmental conditions prevailing. According to the European soil database (ESB, 1998), Cambisols are the dominant soil type (33%), followed by Rendsinas (21%), Podzols (13%), Luvisols (11%), Planosols (7%) and Chernozems (7%). In general, each soil type may be affected by soil erosion. However, the loess soils of the federal province of Lower Austria are especially prone to soil erosion by water. Land management is certainly the key factor for onsite erosion risk. As a consequence of the great extent of alpine territory, forests (3 260 000 ha) and grassland (1 917 000 ha) both cover larger areas than arable land (1 382 000 ha). Although areas covered by forests and grassland do erode, the amounts of soil loss are generally low compared with soil erosion rates that may occur on arable land.

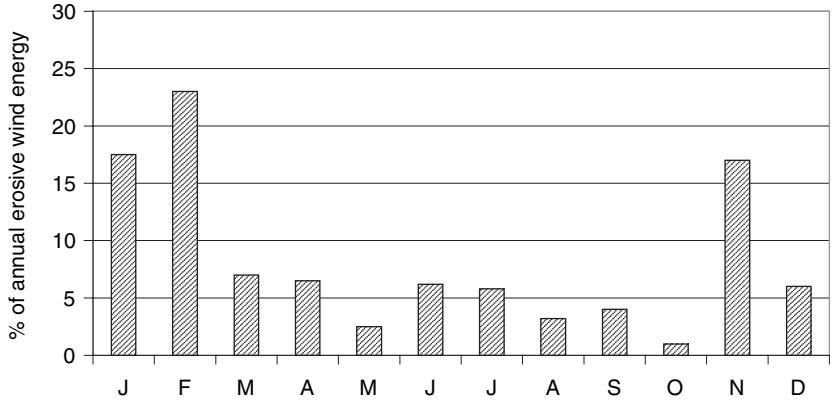

Figure 1.17.2 Seasonal distribution of erosive wind energy for the station Obersiebenbrunn. (After Klik, 2004. Reproduced by permission of A. Klik)

However, a comparison of sediment loads in sub-watersheds of the Ybbs river (province of Lower Austria) with different land use gave similar sediment loads of 0.4 t ha^{-1} yr^{-1} for both an alpine watershed without remarkable agricultural use and a pre-alpine watershed under intensive agricultural use. A simple reason for this was the huge difference in total flow rates, indicating that the key factors of erosion may have different importance for on- and off-site effects.

1.17.3 AMOUNTS OF SOIL EROSION BY WATER – MEANS AND EXTREMES

No efforts have been undertaken so far to estimate historical soil loss rates quantitatively, but the qualitative evidence of historical soil erosion exists in various forms. One of them is the formation of large gullies on loess soils of Lower Austria, which can be found especially in the wine-growing region of the 'Wagram', a distinct terrace landscape feature of the River Danube. Nowadays, these gullies are partly used as agricultural roads ('Hohlweg'). Truncation and accumulation of soil profiles can be observed fairly commonly. Strauss and Klaghofer (2001) described soil profiles and found huge variations for the rootable soil depths at different positions on a catena ranging from 25 cm (shoulder) to 130 cm (footslope). These differences are the result of a combined effect of water erosion and tillage translocation.

As no measured data on soil loss due to wind erosion are available, we only report results obtained in experiments to estimate soil erosion by water. Experiments in general are focusing on effects of different management practices on soil loss. Klik (2003) investigated the effects of different tillage practices on soil loss, runoff, nutrient and pesticide losses on three different sites in Lower Austria. He measured mean soil loss rates over 9 years of between 5 and 39 t ha^{-1} yr^{-1} for conventional plots compared with 2–6 t ha^{-1} yr^{-1} for the plots where mulching was applied and 0.5–4 t ha^{-1} yr^{-1} on plots with direct drilling. Kunisch *et al.* (1997) compared the effects of different machinery (plough, rotary cultivator) and different conservation practices (conventional, mulching, direct drilling) on soil erosion. Mean soil loss rates (2–3 years) for conventional plots were 6 t ha^{-1} yr^{-1} whereas for all other treatments soil loss rates were below 1 t ha^{-1} yr^{-1}. Pollhammer (1997) compared the effects of different machinery on soil loss at two different sites in Styria. He reported mean (1–2 years) soil loss rates of between 8 and 72 t ha^{-1} yr^{-1} for ploughed plots compared with soil loss between 1 and 46 t ha^{-1} yr^{-1} for chiseled plots. However, the results were greatly influenced by one extreme event. Soil loss for this event with a total rainfall amount of 64 mm was measured at 55 t ha^{-1} yr^{-1}. Another example of the effect of extreme events was given by Strauss and Klaghofer (2004). They mapped linear soil features within a small experimental watershed after a 5-day period of heavy rain (115 mm) and recorded total amounts of soil transport by rilling of more than 730 t in a watershed of 2.89 km^2. Only a few fields contributed to this amount, the highest soil loss being recorded in one field with a total soil loss of almost 300 t ha^{-1}. However, the amounts of sediment which left the watershed during this event were only about 17 t, most of the eroded material being redistributed within the watershed. Similarly to the high inter-annual variations in summer erosion, tremendous differences in the actual amounts of soil loss due to winter erosion by snowmelt can be observed. Unfortunately, there is not much quantitative information available to report on long-term experiments. However, available results suggest that winter erosion is a widespread phenomenon. Gerlich (1997) reported soil losses during snowmelt of between 2.1 and 5.1 t ha^{-1} for a one winter experiment. Schönhart-Klenkhart (1986), in a 2-year experiment, measured phosphorus losses (as a surrogate for soil loss) during three different periods during the year. Whereas the phosphorus loss was negligible for the period January–April during the first year, it became the dominant erosion period during the second year owing to a snowmelt event.

Owing to their rough environment, alpine soils become increasingly fragile with increasing height above sea level and their capability to regenerate decreases. This is of special importance when the soil-protecting cover becomes destroyed or removed. Construction of ski runs is a long-lasting problem in that context (Tappeiner,

1988). Krautzer *et al.* (2003), after testing the protection effectiveness of different measures to restore vegetation on ski runs, concluded that, independently of the various techniques of seeding, an additional cover of the top soil (straw, hay or similar) is necessary to reduce soil erosion and effectively restore ski runs at high altitudes. On the other hand, abandonment of cultivated areas in alpine regions may cause an additional threat of landslides. Within a set of 12 factors responsible for increasing landslide risk, Tasser *et al.* (2003) identified reduced land management as a main cause of an increased landslide risk.

1.17.4 AREAS AFFECTED BY WATER EROSION

Since the work of Wischmeier and Smith (1978), we have been aware of the importance of the different factors that influence the level of soil erosion at field scale. The basic assumption was that land use is the most important single feature to accelerate soil erosion by water. Therefore, information about the spatial distribution of land use should provide basic information about the spatial extent of erosion risk areas in Austria. Land use information was obtained from the Austrian agricultural statistics data for the year 1999. They provide information about the extent of the different types of agricultural land use at the level of communities. We considered maize, sugar beet, potatoes, vineyards and orchards as potentially subject to erosion by water. For each community, these areas were added up and the potential erosion risk was expressed as a percentage compared with the total agriculturally used land. As a result, Figure 1.17.3 gives

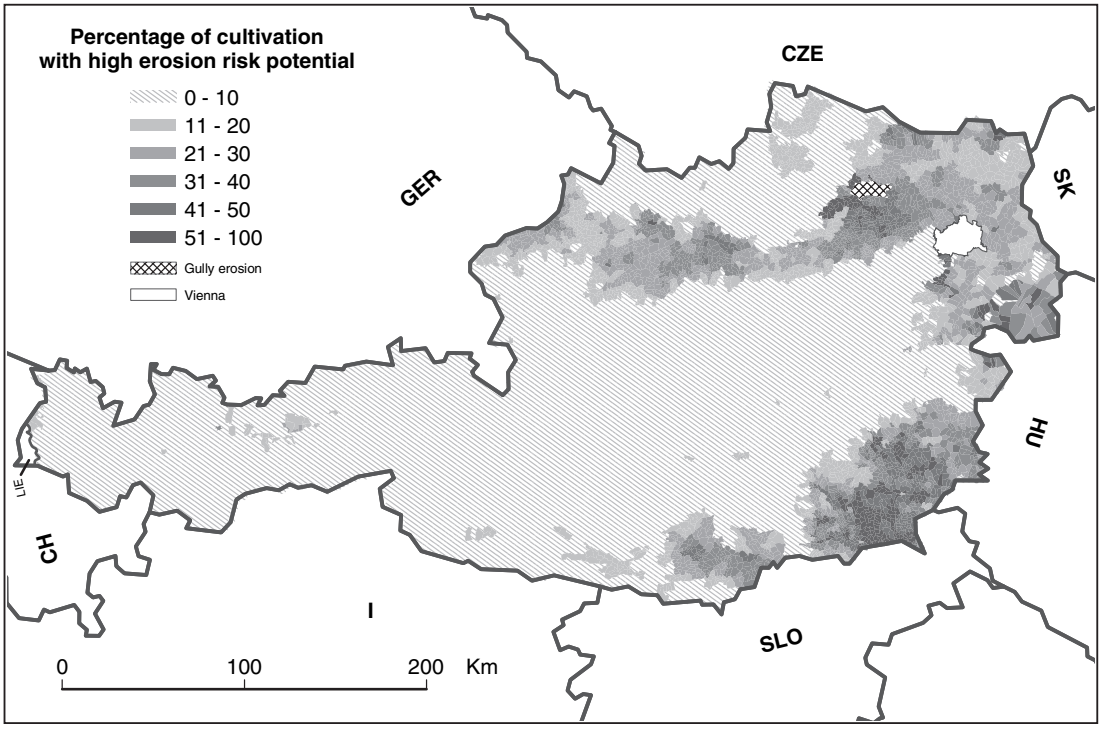

Figure 1.17.3 Percentage of cultivation with potential erosion risk, aggregated at the level of Austrian communities and areas with gully erosion

an overview of areas within Austria, in 1999, at risk of erosion. It includes no information about the amount of actual soil loss which can be expected. In a further attempt, we also included mean slope at community level as a major factor for erosion. Although the general distribution of erosion risk within Austria did not change a lot, for some communities unrealistic results were produced. This happened, for instance, in communities with a high degree of variation in slope such as communities in narrow alpine valleys where agricultural land is situated on the valley floor on low slopes but the communities also include the adjacent steep slopes. On the other hand, for a few regions, especially for communities in basins such as 'Tullner Feld' or 'Marchfeld', exclusion of slopes may also lead to misinterpretations. No attempt was made at this stage to include additional factors (soils, climate) or to weight the results and create some kind of risk index. In the case of soil properties, no appropriate Austrian soil information is available at present on the soil database for Europe (ESB, 1998), because its spatial resolution is not suited to be applied at the community level (Strauss and Wolkerstorfer, 2003).

In 1999, a total area of 439 300 ha under crops was potentially subject to erosion. This amounts to 13% of the total area of agricultural land (3 381 000 ha) if woodland is not included. Inclusion of woodland (3 260 000 ha in 1999) and others (1 747 086 ha in 1999) leads to an extent of 5.2% of the Austrian territory with a potential high erosion risk. The spatial distribution of potential erosion risk is very heterogeneous. The main affected areas include the productive areas of the southeast and northeast plains and hills, the Alpine foreland and the Carinthian basin.

1.17.5 AREAS AFFECTED BY WIND EROSION

Soil erosion by wind mainly occurs within the great basins of Eastern Austria. The spatial distribution of soils which are susceptible to wind erosion is given in Figure 1.17.4. Two different types of soils at risk can be observed. Parts of this area are covered with sands, but special care needs also to be taken in the case of so-called 'Feuchtschwarzerden', which belong to the soil group of Chernozems. These are soils that were influenced formerly by high groundwater tables. In the case of drying, they become very susceptible to wind erosion owing to their high content of organic matter with a very low specific weight (type B in Figure 1.17.4). Areas covered by sands have been recognized as risk zones since the 18th century. Already in 1770, the Empress Maria Theresia ordered reafforestation of parts of this zone to stop shifting sand dunes. This is one of the first known attempts at lowland reaforrestation (Wendelberger, 1955). Owing to early recognition of the wind erosion problem in the sandy areas, they are now almost stabilized (type A in Figure 1.17.4).

Mainly in the months of November, January and February, higher wind speeds can be observed which, in combination with an absence of a protective land cover and soils that are prone to wind erosion, leads to soil losses by wind. An Austria-wide mapping of wind erosion risk does not exist at present. Klik (2004) estimated soil loss by wind for the 'Marchfeld', an intensively used plain east of Vienna using the WEQ (Woodruff and Siddoway, 1965). Calculated average erosion rates from single fields ranged from 0 to 5.4 $t\,ha^{-1}\,yr^{-1}$ and indicate low to medium erosion risk. Unfortunately, no measurements are available to validate these results.

1.17.6 EROSION CONTROL MEASURES

1.17.6.1 Water Erosion

With the participation of Austria in the European Union, the first concerted efforts to reduce soil erosion by water at national scale started. The Austrian programme for a sustainable agriculture (ÖPUL) was launched

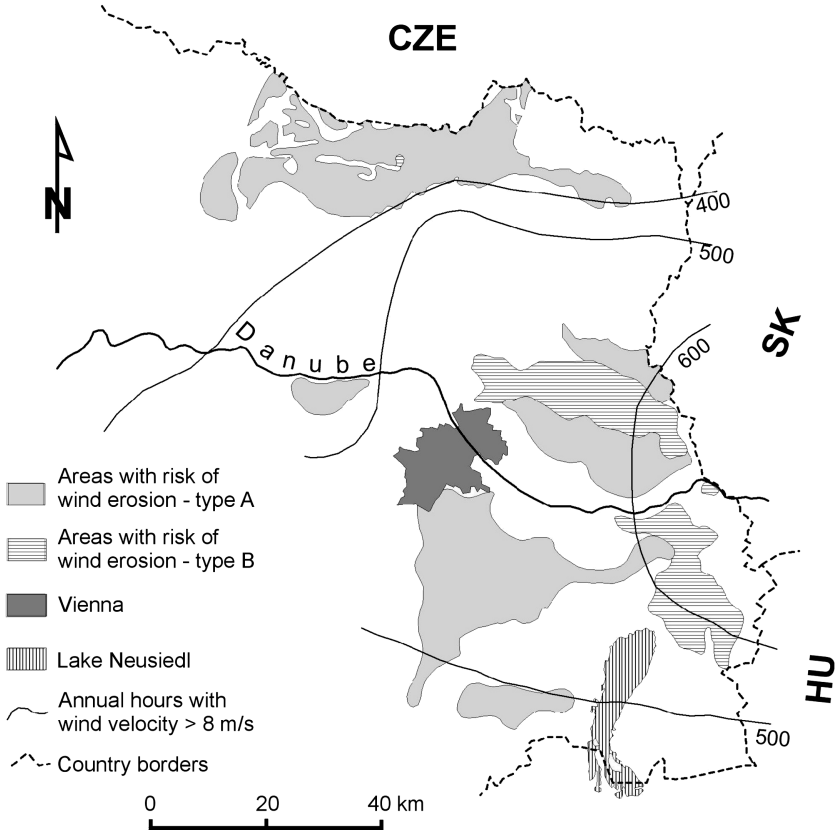

Figure 1.17.4 Map of soils which are susceptible to wind erosion (for explanation of types A and B, see text) for the provinces of Lower Austria and Burgenland. After Nowak, 1972, modified. (Reproduced from *Österreich in Geschichte und Literatur* 1972, **16**: 389–401, with permission of Institut für Osterreichkunde)

in 1995. It offers environmental contracts to farmers who are willing to implement specific protection measures such as (I) soil erosion control in vineyards, (II) soil erosion control in orchards and (III) soil erosion control on farmland. In its actual form (BMLFUW, 2000) the main measures in these contracts are as follows:

- *Soil erosion control in vineyards* Covering the soil using either mulching, straw or cover crops between each row from 1 November to 30 April, or terracing.
- *Soil erosion control in orchards* Covering the soil using either mulching, straw or cover crops between each row for at least 10 months per year, or terracing.
- *Soil erosion control in farmland* Conservation tillage (either direct drilling or mulching).

In addition to these contracts, some erosion control effects may also be expected as a result of contracts for the growing of cover crops (winter erosion) or landscape restructuring which are not directly for erosion control reasons.

An evaluation of participation in soil erosion control contracts reveals an increasing trend from 1998 to 2002 (Table 1.17.1). In 2002, an area of about 150 000 ha was under contract. As only cultivation with a potential high erosion risk is subject to the offered control measures, we can conclude that about 34% of the area with

TABLE 1.17.1 Participation (ha) in soil erosion control measures offered by the Austrian environmental programme for a sustainable agriculture (ÖPUL 98 and ÖPUL 2000)

Main production zone (areas in ha)	1998	2001	2002
High alpine area		42	49
Subalpine area	119	212	194
Eastern fringe of the Alps	732	1177	1307
Wald- and Muehlviertel	42	3356	3204
Carinthian basin	11	326	508
Alpine foreland	258	32538	32485
Southeastern area of plains and hills	7019	10894	11029
Northeastern area of plains and hills	1149	94539	101186
Austria	**9336**	**143083**	**150035**

higher potential erosion risk was affected. These results can certainly still be improved. Major drawbacks that farmers see in implementing these practices are the lack of adequate machinery and the additional work load and organization (Seemann, 2003). An amount between €93 and €113 ha^{-1} (different options exist) is given as a subsidy for implementation of the measure 'soil erosion control in farmland', between €145 and €291 ha^{-1} (depending on slope) is paid for measures to control soil erosion in orchards and between €145 and €799 ha^{-1} (again depending on slope of the area) is paid for erosion control measures in vineyards. Adding up mean values for these contracts gives an amount of about €143 million, which was invested in 2001 in measures to reduce soil erosion risk on agricultural land.

1.17.6.2 Wind Erosion

Protection measures against wind erosion have been implemented since the late 1950s mainly in the federal provinces of Lower Austria and Burgenland, which are most affected by the problem (Figure 1.17.4). Since then, only in Lower Austria have windbreaks of a total length of about 2300 km been planted which protect an area of about 100 000 ha. An annual increase of this area of 2500 ha is predicted (Köchl, 2001).

1.17.7 LEGISLATIVE BACKGROUND

The protection of agriculturally used soils is defined in the soil protection laws of the different Austrian provinces. In these laws, the aim of protection, the 'maintenance of a natural soil fertility and of an ecological functioning of soils', is defined. Additionally, the way to achieve this aim is defined: a particular conservation measure may be put into practice. However, no explicit rule is included as to what extent of soil loss is tolerable. Because of this lack of a definition of tolerable soil loss, in general no intolerable soil losses – in terms of legislation – are recognized (Klaghofer, 2002).

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1.18

Germany

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1.18.1 INTRODUCTION

Germany is 357 031 km^2 in size. It can be subdivided into nine large landscape units (Figure 1.18.1), with contrasting natural and anthropogenic erosion conditions (Table 1.18.1). Since World War II, German agriculture has been increasingly competitive based on industrial production-type methods. Subsidies were coupled to production intensity. This led to high-intensity agriculture with several adverse environmental effects, erosion being one of them. Although this has been changing since the 1990s, agriculture is still and for a long time will be shaped by this 50-year-old paradigm.

1.18.1.1 Soil Use

Land use comprises 37% arable land, 17% grassland and 30% forests, the remainder being urban areas and water (Destatis, 2002). To illustrate the relative erosion potential of different crops and rotations under German growing conditions, the *C* factors according to the USLE and the respective acreage are given in Table 1.18.2. With respect to erosion, maize is by far the most important crop considering the acreage and *C* factor of conventional maize rotations. While the percentage of maize was close to zero before the 1960s, maize is now the fourth most important crop, covering 9.4% of the arable land after winter wheat (25.0%), winter barley (11.6%) and rape (10.7%) (numbers for 2002). On the other hand, ley-based rotations disappeared except under organic farming. The most erosion-prone crops, although they cover only small areas, are hops, some vegetables (onion, cucumber) and vines. Hops are concentrated in some parts of the Tertiary Hill Land north of Munich, where the largest hop-growing area in the world can be found. Vines are

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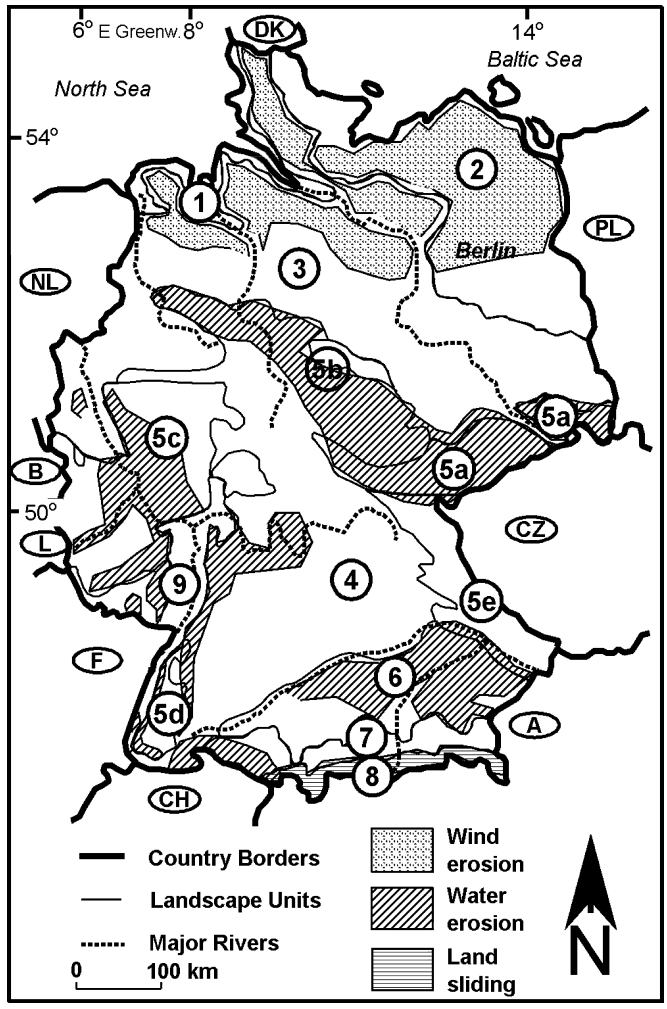

Figure 1.18.1 Areas of severe erosion in Germany. Water erosion shown for soil losses $>10\,t\,ha^{-1}\,yr^{-1}$ and wind erosion for soil losses $>1\,t\,ha^{-1}\,yr^{-1}$ each averaged over agricultural land. The map is based on Auerswald and Schmidt (1986), Deumlich *et al.* (1997), Elhaus (1998), Funk and Frielinghaus (1997), Gündra *et al.* (1995) and Gryschko (2000). Landscape units following Ahnert (1989): 1, Marsh Land; 2, Northern Young (Weichsel) Moraines; 3, Northern Old (Saale) Moraines and Northern Loess Belt; 4, Mesozoic Scarpland; 5, Mountain Ridges: (a) Ore Mountains and Thuringian Forest; (b) Harz; (c) Rhenish Slate Mountains; (d) Black Forest; (e) Bohemian Forest; 6, Tertiary Hill Land; 7, Southern Young (Wurmian) Moraines; 8, German Alps; 9, Upper Rhine valley

mainly concentrated along the River Rhine and its tributaries (Main, Mosel, Saar). Vegetables are traditionally concentrated around large cities (Hamburg, Nürnberg), often on sandy soils and flat terrain. Improvements in transport initiated the move of these high-value cash crops to better, loessial soils, which often are also more undulating. Furthermore, cultivation changed from small-plot gardening type to large-plot agro-industrial type with high axle loads. The contribution of these vegetables to total soil loss is thus increasing.

TABLE 1.18.1 Major landscapes (Figure 1.18.1) and typical values of landscape properties relevant to soil erosion by water, wind and sugar beet harvesting

Land-scape No.	Precipitation ($mm\ yr^{-1}$)	R factor ($N\ h^{-1}\ yr^{-1}$)	Typical texture[a]	Typical K factor ($t\ h\ N^{-1}\ ha^{-1}$)	Relief energy ($m\ km^{-2}$)	Mean wind speed ($m\ s^{-1}$)	Major land use (% total land)			Sugar beet (% arable land)
							Arable	Grass	Forests + urban areas	
1	700–800	40–60	L	0.5	0	5.5	50	50	0	1
2	500–700	35–50	S-L	0.15–0.45	0–25	3.5	50	20	30	3
3	450–800	40–60	lS-sL	0.4–0.5	0–25	4.0	40	10	50	12
4	700–900	50–70	lS-sT	0.35	50–100	2.5	40	30	30	3
5	900–1300	50–110	xsL	0.3	75–200	4.0	20	30	50	1
6	650–850	70–80	sL-L	0.3–0.5	25–75	2.5	60	20	20	8
7	800–1200	80–100	xL	0.3–0.4	25–75	3.5	20	40	40	0
8	1200–2000	110–130	xsL	0.3	>150	3.5	0	50	50	0
9	500–600	50	sL	0.6	0–25	2.5	60	20	20	12

[a]Soil texture: s = sandy, S = sand, l = loamy, L = loam, x = stony, T = clay.

Precipitation and *R* factor from Sauerborn (1994); texture from Bodenschätzung; *K* estimated from Bodenschätzung according to Auerswald (1986); land use from Destatis (2002); relief energy from Richter (1965).

TABLE 1.18.2 Range of *C* factors for arable crops and important crop rotations and cultivated area in 2002 (Destatis, 2002)

Crop/rotation	Cultivated area (km^2)	*C* factor	Reference[a]
Hops	183	0.42–0.78	A03
Vines	1044	0.30–0.59	A99
Rotations:			
33% onion + 67% small grain	72 (onion)	0.21–0.26	A98
33% cucumber + 67% small grain	32 (cucumber)	0.21	A98
33% maize, conv., + 67% small grain	15290 (maize)	0.14–0.18	A03
33% maize, mulched, + 67% small grain		0.05–0.08	A03
33% beet, conv., + 67% small grain	4610 (beet)	0.10–0.14	A03
33% beet, mulched, + 67% small grain		0.05–0.08	A03
Small grain only (including rapeseed)	80700	0.05–0.10	A03

[a]References: A03 = Auerswald *et al.* (2003b); A99 = Auerswald and Schwab (1999); A98 = Auerswald and Kainz (1998)

1.18.1.2 Land Use and Management Changes

Although average land use remains almost constant, there are large changes within Germany. These occur as explained for vegetables, but also for other crops, leading to rather dynamic changes in the influence of land use on soil loss. As an example, the county averages of the *C* factors in Bavaria from 1986 and from 2001 are poorly correlated ($r^2 = 0.19$; Auerswaldk, unpublished). Large changes in land use followed the political changes in eastern Germany after 1989.

In general, there is a trend towards the increased application of mulch tillage (Kainz, 1989) as a control on erosion since this measure is encouraged in some federal states, e.g. in Bavaria the application of mulch tillage increased within a few years to 18.5% of the arable land in 2001 since €100 $ha^{-1}\,yr^{-1}$ is paid for this environmental service. This was the main reason why soil losses decreased on average by 40% between 1986 and 2001 (Auerswald *et al.*, 2003b) but it increased again since this incentive was cut due to EU regulation. The introduction of a Federal Soil Protection Act in 1999 should lead to activities by the administration and by the farmers which foster this trend (Frielinghaus *et al.*, 2002).

1.18.2 WATER AND WIND EROSIVITY

In Germany, continentality increases from the north-west to the south-east. In consequence, the frequency and severity of thunderstorms and the concentration of precipitation during summer months increase along this gradient and cause an increase in rain erosivity. This general trend is modified and further aggravated by orography, which induces an increase in precipitation from the flat lowlands in northern landscapes (ca 500–800 mm yr^{-1}) to the mountain ridges in the centre with 800–1200 mm yr^{-1} and finally to the Alps in the South where precipitation peaks at more than 2000 mm yr^{-1}. Hence rainfall erosivity increases from 40 N $h^{-1}\,yr^{-1}$ in the north-west to 100 N $h^{-1}\,yr^{-1}$ in the south and even exceeds this value in the German Alps. Low precipitation and the lowest rain erosivity can be found in the lee of the mountain ridges (e.g. south-east of the Harz).

Increasing continentality and height above sea level also cause lower winter temperatures and thus more ground frost and snow accumulation. Snowmelt erosion hence increases similarly to rainfall erosivity. In areas of pronounced continentality, Horton-type runoff by intensive thunderstorms and snowmelt runoff prevail whereas in areas of low continentality saturated runoff during winter months is more frequent.

Wind speed (Table 1.18.1) and wind erosivity are greatest close to the coast and decrease inland with higher values at mountain ridges. On the Northern Moraines (Figure 1.18.1) high wind speeds are also found close to

the soil surface owing to the flat topography and cause significant wind erosion. The southern border of the Older Northern Moraines is formed by mountain ridges ('Mittelgebirgsschwelle') with a sudden increase in aerodynamic resistance causing a corresponding decrease in wind erosivity.

1.18.3 SOIL ERODIBILITY, TOPOGRAPHY AND DOMINANT EROSION PROCESSES

The nine landscape units differ greatly in soils, topography and land use (Table 1.18.1). In consequence, the extent and the dominant processes and forms of erosion differ between these landscapes.

1.18.3.1 Marsh Land

The Marsh Land is the area of presumably least erosion problems despite intensive soil use. Fine textured soils, high in organic matter, and a high water table inhibit wind erosion in spite of the proximity to the sea and thus high wind velocities.

1.18.3.2 Northern Young Moraines (Weichsel Glaciation)

In this landscape, topography is characterized by gently undulating relief. Large fields were created especially during the period of the German Democratic Republic (GDR), when agricultural policy followed the paradigm of industrialization. On average, field sizes in eastern Germany are about 50 ha (Frielinghaus, 1998), which is much larger than in the western part (e.g. in Bavaria in 1999: 1.5 ha on arable land). Within these fields, obstacles along the flow path are missing. Hence the runoff from large areas can converge along thalwegs. The sandy soils with clay contents typically below 15% give only limited structural stability and resistance to the hydrodynamic forces of the runoff. In consequence, severe ephemeral gullies form along the thalwegs and frequently exceed soil losses by sheet erosion.

Kettle holes 0.1–5 ha in size with a density of 0.6–40 km^{-2} covering up to 5% of the arable land are a unique feature of this landscape (Kalettka *et al.*, 2001). The high biotic value of these small wet spots is endangered by soil erosion because all sediment is trapped there. The kettle holes also offer a unique chance to quantify overall past erosion by constructing a budget of erosion and deposition (Frielinghaus and Vahrson, 1998). These calculations show that erosion is moderate owing to the comparatively flat terrain, the sandy soils and the low rain erosivity (Table 1.18.1).

The Young and also the Old Moraines in Northern Germany are the only landscapes where water and wind erosion both contribute significantly to soil degradation. Although each process in isolation is only moderate, the combined action has led to a substantial soil loss. For the states of Brandenburg and Mecklenburg-Vorpommern, it was estimated that 16% of the agricultural land is already significantly degraded by water erosion and 8% by wind erosion (BUNR, 2002).

1.18.3.2 Northern Old Moraines (Saale Glaciation)

The Northern Old Moraines are mostly covered either by a coarse-grained loess (sandy loess) or by rather poor sands in areas from which the loess originates. Owing to the higher silt content, the lower stone content and the more intensive land use, the soils are more prone to erosion than those of the Young Moraines. On the other hand, terrain is flatter in general and this results in soil losses comparable to those of the Young Moraines. Again, field layout and soil properties lead to the coexistence of moderate sheet and rill erosion and severe ephemeral gullying along the thalwegs. In contrast to the Young Moraines, kettle holes are missing.

Drainage is generally good where sandy fluvial sediments instead of loess cover the Old Moraines. These well-drained areas are characterized by wind erosion. Under the given climatic conditions, saturated runoff may cause some water erosion during winter months.

1.18.3.4 Mesozoic Scarpland

The Mesozoic sediments feature contrasting properties in close proximity. Hard limestone rocks, calcareous or acidic clays, sandstones of various size classes cemented by silica, iron oxides or carbonates exist, giving rise to poor soils that often have deficits in soil chemical and physical aspects. Pleistocene solifluction and Holocene erosion helped to create superficial sediments which are more suitable to plants because the mixing of the contrasting materials created soils of loamy texture and more balanced chemical properties (Auerswald *et al.*, 1991).

Topographically, the Mesozoic Scarpland exhibits typical cuesta features of steep scarp faces where slowly weathering rocks outcrop while the softer parent rocks, especially clays, comprise the dip slopes. Arable land use is mostly restricted to this flatter land whereas the scarp faces are forested. Hence soil erosion on the arable land is mostly lower than $5\ t\ ha^{-1}\ yr^{-1}$. Considering also the forested land, soil erosion mostly remains below $3\ t\ ha^{-1}\ yr^{-1}$. Nevertheless, soil erosion is threatening these soils because the layer of solifluction and Pleistocene weathering of the hard rocks is mostly shallow (<1 m). High losses occur, where the Mesozoic sediments are covered by loess (e.g. in the Ochsenfurter Gäu or in the Kraichgau between Heidelberg and Pforzheim) because of the high erodibility of the loess and the intensive arable use (Auerswald and Schmidt, 1986; Dikau, 1986; Gündra *et al.*, 1995).

1.18.3.5 Mountain Ridges

The Mountain Ridges are characterized by high rainfall erosivity and steep gradients. In spite of the resulting high erosion potential, actual erosion remains low because forests and grass land prevail (Voss, 1978). Arable land use in these areas traditionally was characterized by an above-average percentage of rotations containing grass/legume leys with comparable low *C* factors owing to the years under grass and their carry-over effect to the following years. At lower altitudes, a mild climate and soils often containing significant amounts of loessial material allowed for more intensive arable use with a high row crop percentage. The Mountain Ridges are therefore often surrounded by a fringe of high erosion because the high-intensity agriculture of the lowlands expands into an area of high natural erosion potential. This was found for the Rhenish Slate Mountains (Botschek, 1999), the Harz (Seils, 2000), the Bohemian Forest (Auerswald and Schmidt, 1986), the Ore Mountains and the Thuringian Forest.

Where the loessial and hence easily dispersible soil is combined with grassland use, tunnel erosion can occur. This has frequently been found in the Rhenish Slate Mountains (landscape 5c in Figure 1.18.1), where more than 50% of all reported tunnel erosion sites are located (Winzen and Botschek, 2003). However, even in this landscape tunnel erosion is far less than surface erosion.

During the second half of the 20th century, land use has faced considerable changes in these landscapes. Farmers earn most of their money with other jobs, often in cities, at a great distance from their farms. They replaced labour-intensive grassland and arable leys with maize, which depends less than hay making on the weather and can therefore more easily be brought in line with the industrial jobs. This replacement of low-erosion crops (grassland, arable leys) by high-erosion crops (maize) especially in landscapes which, owing to their low temperatures, do not favour maize growth, considerably increased erosion. This was further intensified by simultaneous changes in the rather small-parcelled field layout to larger units in order to decrease labour demand. The loss of hedges, field borders and lynchets, the increase in effective slope length and the more homogeneous land use further intensified the erosion problems. These changes were more

pronounced in the area of the former GDR owing to the development of industrialized agricultural systems. Hence very high erosion rates can be found in the Ore Mountains, the Thuringian Forest and the Harz Foreland (Deumlich *et al.*, 1997; Seils, 2000). Moreover, owing to the high rainfall and rather sandy soils of low structural stability, thalweg erosion has become a frequent phenomenon.

Recent developments go in the opposite direction. Labour availability in agriculture has become so limited that the labour-intensive dairy cows have been replaced. Arable land is increasingly grazed by suckler cows or even replaced by a succession of grassland, bush land and forests. This is encouraged by payments for environmental services. Soil erosion has declined considerably where this is the case.

1.18.3.6 Tertiary Hill Land

Unconsolidated Tertiary sediments, mostly gravelly sands, are covered by a thin (0–3 m) layer of loess, leading to fertile soils and intensive arable use. The terrain is characterized by asymmetric valleys where the steep west-facing slopes (~30%) are forested, the valley bottoms are under grassland and the gentler east-facing slopes(~10%) are under arable use. Land reconsolidation has created fields where the erosive slope length extends over several hundred metres. In consequence, high rates of water erosion occur, often around $10\,t\,ha^{-1}\,yr^{-1}$. This soil loss endangers soil fertility because the underlying quartz sand offers a very low fertility and hence fertility declines rapidly with the loss of the shallow loess cover (Stürmer *et al.*, 1982).

1.18.3.7 Southern Young Moraines (Wurm Glaciation)

High precipitation ($900–1100\,mm\,yr^{-1}$) and high rain erosivity ($80–100\,N\,h^{-1}\,yr^{-1}$) are due to orographic rains along the Alps. On the other hand, soil erodibility is low ($K < 0.20\,t\,h\,N^{-}1\,ha^{-1}$) because even arable soils are often rich in organic matter (4–8%), clay (30% of the fine earth) and rock fragments (10–30%). The terrain is much steeper than on the northern moraines, with slopes frequently exceeding 20%. Land use is mainly grassland, because of the high rainfall, leading to generally low erosion rates. Intensification of agricultural production during the last four decades of the 20th century has increased the demand for a high-energy supplement in addition to the grass forage high in protein. This demand was met by the cultivation of maize. Although this is an old crop in this landscape, its cultivation has changed dramatically after the development of herbicides. Beforehand, maize was cultivated for only a few years until weeds became abundant. The arable field was returned to grassland and another part of the grassland came under arable use for some years. High structural stability and the earthworm channels were acquired during the grassland period. The frequent change in land use distributed the soil loss evenly within the landscape. After the introduction of herbicides, sites more favourable for arable use remained arable and the other sites remained grassland. This decreased organic matter, structural stability and earthworm channels on the permanent arable land. The steeper parts of the landscape are preferred for arable use because of their better drainage whereas the depressions, but also the crests, often carry wet, even peat soils which are preferred for grassland. This change in land use has thus increased erosion considerably and created a small-scale mosaic of well-protected grassland and highly erodible arable land on which maize is often cultivated in monoculture.

1.18.3.8 German Alps

In spite of its steep terrain and the high rainfall with high rain erosivity, the Northern Alps mostly exhibit low water erosion rates because the whole land is under forest or grass (Breitsameter, 1995; Johannes, 1995; see Appendix, Table 1.18.A1). Only in areas where humans have destabilized the surface (by building forest and farm roads, by the construction and reshaping of skiing areas), where snow creep has removed the soil cover or

where the retreat of the Pleistocene glaciers has left steep slopes in unconsolidated material do high water erosion rates occur (Becht, 1989; Bunza, 1992; Dommermuth, 1995).

Soil creep is a common phenomenon (Bunza, 1992, Dommermuth, 1995). On grazed land, shallow soil creep is caused by heavy cattle leading to typical cattle steps (terracettes). Where clayey sediments prevail, especially in the 'Flysch' region, deeper soil creep (rotational creep) can occur. These areas are mostly forested, with the forest being protected by law ('Schutzwald').

1.18.3.9 Rhine Rift Valley

The flat terrain, the low rainfall erosivity and the low wind speed are responsible for very low erosion rates. Soil losses mainly result from the harvest of sugar beet, which constitutes an important crop in this landscape. Only at the fringe of neighbouring landscapes can high erosion losses by rain result from the pronounced increase in slope gradient in combination with the high erodibility, intensive arable use of the loessial soils and the cultivation of vines.

1.18.4 DAMAGE AND COSTS

1.18.4.1 Total Accumulated Soil Loss

Soil truncation mapping, which includes the integral action of all erosion processes since the onset of land use, indicates that large parts of Germany are already considerably damaged by erosion. In the Tertiary Hill Land (Maier and Schwertmann, 1981) and the Rhenish Slate Mountains (Richter, 1987), only 18% of the total mapped area was little or not affected by erosion. In the Northern Young moraines, accumulated erosion damage seems to be less and the areas of erosion and deposition are more or less balanced, indicating a closed budget within this landscape (Schmidt, 1991; Armanto, 1992).

1.18.4.2 Harvest Erosion

Sugar beet production is concentrated around sugar factories on mostly flat loessial areas (for distribution among landscapes, see Table 1.18.1). Average losses by soil adhering to sugar beet typically range between 5 and 8 t ha^{-1} yr^{-1} with large annual fluctuations depending on weather conditions during the beet harvest. In general, harvest erosion is about 40% greater in northern than southern Germany (Table 1.18.3) in spite of the higher clay content in the soils of the south. This is due to the more pronounced continentality in southern Germany, which causes weather conditions during harvest that are favourable for a better soil–beet separation.

TABLE 1.18.3 Soil losses from sugar beet harvesting as recorded at the sugar factories (Schulze Lammers, unpublished)

	Percentage of soil delivered with beet (%)				Area (ha)	Harvest loss (t ha^{-1} yr^{-1})
Year	Schleswig-Holstein	Nordrhein-Westfalen	Bavaria	Germany	Germany	Germany
1980–1984	22.0	17.7	14.1	17.2	426660	8.4
1985–1989	17.4	15.0	10.9	14.3	395363	7.3
1990–1994	13.3	12.8	11.2	12.8	556513	6.3
1995–1999	11.7	11.3	10.1	10.9	505095	5.7
2000	9.6	11.1	8.4	9.0	451410	5.6
Mean	15.8	14.1	11.4	13.6	469980	6.8

Harvest erosion decreased by about 44% during the last two decades of the 20th century owing to technical improvements in soil–beet separation and because of temporary on-farm storage of the beet in clamps. Whereas the first really decreased losses, the second only reduced soil transport to the factory but still allowed translocation within the farm from the growing to the clamp sites (Schulze Lammers and Strätz, 2003).

1.18.4.3 Water Erosion

An overview of measured recent soil loss rates in the different landscapes (see Appendix, Table 1.18.A1) shows that high soil losses can occur in every landscape. The temporal variability is high. Although Table 1.18.A1 reports average values over several years in most cases, some values are still considerably influenced by single extreme events whereas others seem to be influenced by a lack of large events. Hence a detailed comparison between the different measurements fails. Nevertheless, it becomes clear that land use plays a dominant role (Table 1.18.4). The highest soil losses, close to 60 t ha^{-1} yr^{-1} , were found for hops. Arable soil use mostly produces soil losses between 1 and 20 t ha^{-1} yr^{-1} whereas grassland and forests remain significantly below 1 t ha^{-1} yr^{-1}. It also becomes obvious that with soil conservation systems soil loss even under arable use can be lowered considerably (see especially the results from Scheyern in Table 1.18.A1, which have a wide experimental base). Organic farming, although tested in only one study, seems also to be soil conserving.

Multiplying the data of Table 1.18.4 with the relative acreage of the different crop types in Germany yields an average soil loss on arable land of 4.3 t ha^{-1} yr^{-1}. Landscapes with little or no erosion are underrepresented in this average, although it is based on more than 700 plot years of measurement. On the other hand, the data from Scheyern, which contribute almost one-third of the plot years under annual arable crops, were very low owing to comprehensive soil conservation measures (average without these data: 5.3 t ha^{-1} yr^{-1}). The overall average may therefore be a good approximation. Assuming that the L factor of the USLE is suitable, we can recalculate the data for a erosive slope length of 200 m and a gradient of 9%, which is closer to reality than the plot lengths used in most studies. Recalculation yields an average soil loss of 5.5 t ha^{-1} yr^{-1} for arable land in Germany, 3.9 t ha^{-1} yr^{-1} for all agricultural land and 2.6 t ha^{-1} yr^{-1} if forests are also included.

A comparison of bare fallow soil losses between different landscapes indicates that the Tertiary Hill Land seem to have the highest erosion risk (Table 1.18.5); the much higher value for Northern old moraines is based on a very small data set). This erosion risk includes only differences in soil and rain because data mainly originate from plot studies with slope gradients around 10% and slope lengths < 20 m. Also under arable crops, the Tertiary Hill Land exhibited the highest average soil loss with hops contributing considerably to this value.

TABLE 1.18.4 Influence of land use on sheet and rill erosion rates

Land use	Plot years	Average soil loss (t ha^{-1} yr^{-1})	Adjusted soil loss (t ha^{-1} yr^{-1})
Bare fallow	293.5	21.0	90.8
Hops	222.0	57.9	221.5
Vines	161.0	9.6	3.9
Annual crops	319.5	3.9	4.5
Grassland	9.4	0.14	0.51
Forest	5.4	0.003	0.014
Total	1010.8		

(Data taken from Table 1.18.A1; different studies were combined by weighting them for number of plot years; adjusted soil loss is the measured soil loss adjusted with the USLE LS factor to a uniform gradient of 9% and a uniform slope length of 200 m).

TABLE 1.18.5 Erosion rates in German landscapes

Landscape	Bare fallow		Arable land use	
	Average soil loss ($t\,ha^{-1}\,yr^{-1}$)	Plot years	Average soil loss ($t\,ha^{-1}\,yr^{-1}$)	Plot years
Northern Moraines	13.8	4.8	14.2	14.5
Northern Old Moraines	82.5	1.6	—	—
Mesozoic Scarpland	16.5	163.5	8.0	262.0
Mountain Ridges	15.0	38.6	0.7	57
Tertiary Hill Land	35.8	70.0	38.5	345.0
Southern Young Moraines	11.2	15.0	0.3	24.0
German Alps	—	—	(0.7)[a]	3.2[a]
Average	21.0	293.5	22.6	702.5

[a]mainly pasture (not included in overall average)

Most data are from plot studies and do not include differences in topography, field layout and land use distribution between different landscapes; data were taken from Table 1.18.A1.

1.18.4.4 Tillage Erosion

Sheet and rill erosion rates are mainly reported in Table 1.18.A1. Especially in old studies, using tracers or soil truncation mapping, the soil losses often were wrongly assigned to sheet and rill erosion while tillage or other forms of erosion were overlooked. There are only two studies published, that quantify tillage erosion, both reporting high translocation rates (Table 1.18.6, Figure 1.18.2). Both studies analyse special situations which do not allow extrapolation of the quantities of tillage (and water) erosion to other arable fields. In one study conducted in hop gardens, there were up to 10 tillage operations per year during the study period (Table 1.18.6). The other study examined a very steep field (up to 30%) on which traction allowed some tillage operations only to be performed down slope (Figure 1.18.2). A cautious interpretation of both studies and extrapolation to other sites indicate that tillage erosion often is of a similar order of magnitude to water erosion. This is to be expected from the complexity of topography, the small-parcelled land use in many landscapes and the frequent use of

TABLE 1.18.6 Water and tillage erosion in hop gardens as estimated from long-term copper budgets

Field No.	Number of years	Slope gradient (%)	Slope length (m)	Water erosion ($t\,ha^{-1}\,yr^{-1}$)	Tillage erosion ($t\,ha^{-1}\,yr^{-1}$)	Total erosion ($t\,ha^{-1}\,yr^{-1}$)
1	45	4.7	45	52	28	90
2	34	5.9	65	55	39	94
3	45	2.4	70	15	27	42
4	31	4.5	130	77	42	119
5	22	6.2	210	205	63	268
6	45	4.6	50	24	32	56
Average	37	4.7	95	58	36	96

Values were originally assigned to water erosion only by Schwertmann and Schmidt (1980); they were recalculated and assigned to water and tillage erosion separately using the raw data taken from Schmidt (1979) under the assumption that all deposition within the field is due to tillage erosion.

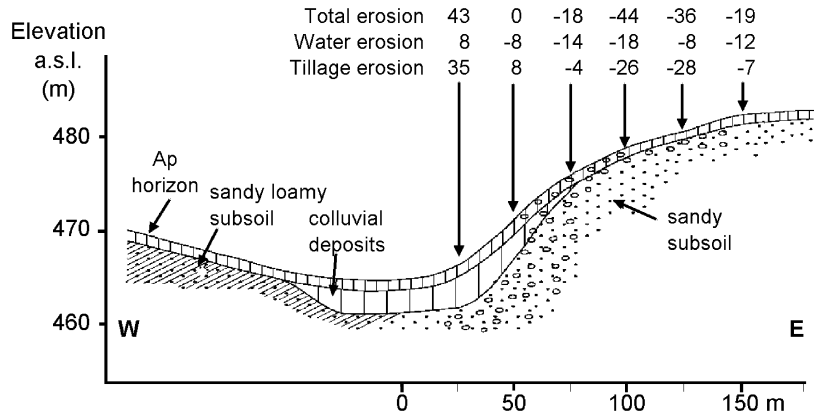

Figure 1.18.2 Average soil translocation (in $t\,ha^{-1}\,yr^{-1}$) of nine transects along a slope quantified by atomic weapon fallout plutonium. Negative values are losses, positive values are gains. Calculated from data of Schimmack *et al.* (2002)

tillage (typically one summer tillage, one autumn ploughing, two secondary tillage passes) and an average plough depth of around 25 cm.

1.18.4.5 Wind Erosion

An overall quantification of wind erosion based on measurements is hardly possible. Few data have been published (e.g. Hassenpflug, 1998; Goossens *et al.*, 2001; Böhner *et al.*, 2003), and they cannot be converted into soil loss per unit area to be comparable with other forms of erosion. Soil transport is highest during spring and autumn owing to the high wind speeds and the sparse soil cover at that time. Tillage seems to play an important role in loosening the soil. Tillage-induced dust emissions were 6.6 times higher than wind-induced dust emissions in Lower Saxony (Goossens *et al.*, 2001).

1.18.4.6 Off-site Damage

In recent years, erosion awareness has increasingly focused on off-site damage, especially the transfer of pesticides (Frede and Fischer, 1997; Müller *et al.*, 2002) and nutrients into down-slope ecosystems. Considerable attention has been given to phosphorus (P) loss (Hamm, 1991; Werner and Wodsak, 1995; Auerswald, 1997), which contributes 31% to the 57 500 $t\,yr^{-1}$ total P input into surface water bodies in Germany (UBA, 1994). This large share results from changes in four different areas (Hamm, 1991; Auerswald, 1997):

1. The surface water bodies, especially large streams, became more sensitive to nutrient inputs because flow velocity was decreased by dams. Surrounding wetlands, which acted as buffer zones, became drained and are often now under arable use. Surrounding trees, which shaded the water surface and thus prevented eutrophication, were removed during the reconstruction of the surface water bodies.
2. Since the 1960s, major efforts have been made to decrease phosphorus input from urban areas (use of P-free detergents; almost complete collection of waste water in sewage plants; improved efficiency of sewage plants due to chemical P precipitation). The decrease in P input from urban areas has increased the relative importance of erosion.

TABLE 1.18.7 Costs for sediment removal

	1988 (DM yr^{-1} ha^{-1} AL)	2003 (€ yr^{-1} ha^{-1} AL)	2003 (million € yr^{-1} for 1.3×10^6 ha AL)
Removal from roads and ditches	7.62	8.10	10.7
Removal from small streams	1.90	2.02	2.7
Removal from large streams	4.29	4.56	6.0
Total			19.4

After Doleschel (1993), modified. Costs expressed in German marks (DM) per hectare of arable land (AL) were converted to euros per hectare of AL and recalculated for 2003 assuming an annual increase in costs of 5%; costs include removal only and not transport and storage.

3. The absolute input from erosion has increased during the same period, mainly caused by an increase in field sizes and the introduction of maize into German agriculture.
4. The P content of the top soils has increased considerably since the 1960s because of a surplus input of P, which amounts to about 1000 kg ha^{-1} of agricultural land averaged over Germany. Although the surplus input decreased from about 30 kg ha^{-1} yr^{-1} in the 1970s to less than 10 kg ha^{-1} yr^{-1} now, there is still a surplus which prevents a decrease in P content. The main share of the surplus in the 1970s originated from mineral fertilizers, whereas now imported animal fodder mainly causes the P surplus of the German agriculture.

1.18.4.8 Costs

On- and off-site damage is manifold and a comprehensive evaluation of all costs is lacking and presumably it is impossible to cover all aspects. Owing to the dense population, infrastructure and industrialization, off-site damages is a major concern. Considering only the removal of erosion sediments from roads, ditches and streams, costs of €19 million yr^{-1} can be expected (Table 1.18.7). If we assume that an amount of P identical with that delivered by erosion to surface water bodies would have to be removed from urban sources by P precipitation (assumed costs €2.4 kg^{-1} P), another €44 million yr^{-1} would be necessary. If however, we use the costs for the removal of 1 kg of P from a lake (€148 kg^{-1} P; Nusch and Brinkmann, 2003), the amount of P delivered by erosion to surface water bodies would lead to costs of €2642 million yr^{-1}. The wide span within the calculated costs of one damaging substance and the diversity of damage illustrates that although a calculation of all costs is impossible, the economic impact is tremendous.

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APPENDIX

Measured soil losses in different landscapes are given in Table 1.18.A1. Note that although the table indicates comparability, this could not fully be achieved owing to missing information, incomplete years of measurement, different approaches, unique situations and a lack of information about the expected return periods of the measured events in relation to the length of observation period. To include studies with measuring periods less than 1 year, average monthly soil losses were calculated and multiplied by 12 to yield annual rates. No corrections were made for differences in slope gradient, plot length and size. Abbreviations used are as follows: *Type of erosion*: S, sheet and rill erosion; T, tillage erosion; E, ephemeral gully; W, wind erosion. *Type of study*: P, plots; M, mapping; T, tracer; W, field, small watersheds, windsheds. *Soil texture*: Sa, sand >0.63 mm; Si, silt; Cl, clay <2 μm.

TABLE 1.18.A1 Measured soil losses in different landscapes

Landscape unit	Location	Land use	Slope (%)	Slope length (m)	Area	Plot years	Type of study	Soil texture Sa/Si/Cl (% bulk soil)	Rainfall (mm yr^{-1})	Mean erosion (t ha^{-1} yr^{-1})	Erosion type	References[a]
2	Dedelow	Bare fallow	11–14	20		2.4	P			21.9	S	F98
2	Dedelow	Row crop	11–14	20		2.4	P			5.0	S	F98
2	Dedelow	Small grain	11–14	20		2.4	P			0.4	S	F98
2	Kiel	Bare fallow	10	5.33	12 m^2	2.3	P	2 soils		5.4	S	G89a,b
2	Kiel	Maize	10	5.33	12 m^2	2.3	P	2 soils		4.6	S	G89a,b
2	Kiel	Maize + clover	10	5.33	12 m^2	2.3	P	2 soils		2.5	S	G89a,b
2	Müncheberg	Row crop	9–10	50		5	P			35.2	S	F98
3	Eschmar	Bare fallow	13.5	10	15 m^2	0.9	P	71/14/10	632	140.6	S	B91
3	Nottuln	Bare fallow	6.0	10	15 m^2	0.7	P	38/37/20	692	7.7	S	B91
4		Bare fallow	9.5	8	8 m^2	60	P	12 soils	725	31.2	S	M88, A93
4	Albacher Hof	Bare fallow	10	8	16 m^2	28.0	P	12/63/25	761	2.2	S	J80
4	Albacher Hof	Arable crops	10	8	16 m^2	12.0	P	12/63/25	761	1.1	S	J80
4	Albacher Hof	Bare fallow	11	8	16 m^2	5	P	12/63/25	761	8.0	S	K56
4	BE2	Barley	21.3	60	720 m^2	1	W	8/74/7	1100	0.7	S	D86
4	Bitbg. Ch.	Spring barley	8	8	8 m^2	3.0	P		765	0.1	S	R87
4	Bitbg. Ch.	Bare fallow	8	8	8 m^2	2.0	P		765	1.8	S	R87
4	Blumberg	Bare fallow	15	8	16 m^2	6.0	P	37/25/12	809	3.6	S	J80
4	Blumberg	Arable crops	15	8	16 m^2	9.0	P	37/25/12	809	0.8	S	J80
4	Dickes Kreuz	Spring barley	8	8	8 m^2	3.0	P		1042	0.0	S	R87
4	Dickes Kreuz	Bare fallow	8	8	8 m^2	0.5	P		1042	2.6	S	R87
4	EH1	Maize	17.6	40	4700 m^2	1	W	8/81/10	1100	36	S	D86
4	EH1	Sugar beet	17.6	60	6100 m^2	1	W	8/81/10	1100	458	S	D86
4	EH1	Oat	17.6	60	7000 m^2	1	W	8/81/10	1100	1.5	S	D86
4	Hollmuth	Bare fallow	23.3	5	10 m^2	3	P	17/75/14	885	1.7	S	D86
4	Hollmuth	Bare fallow	23.3	2	4 m^2	3	P	17/75/14	885	49.8	S	D86
4	Hollmuth	Bare fallow	23.3	2	4 m^2	3	P	17/75/14	885	33.8	S	D86
4	Hollmuth	Bare fallow	23.5	5	10 m^2	3	P	17/75/14	885	37.2	S	D86
4	Hollmuth	Bare fallow	22.2	10	20 m^2	3	P	17/75/14	885	19.7	S	D86
4	Hollmuth	Bare fallow	22.2	20	40 m^2	3	P	17/75/14	885	18.3	S	D86
4	Hungelsberg	Spring barley	8	8	8 m^2	3.0	P		775	0.1	S	R87
4	Hungelsberg	Bare fallow	8	8	8 m^2	2.0	P		775	3.2	S	R87
4	Kockelsberg	Spring barley	8	8	8 m^2	3.0	P		718	0.1	S	R87
4	Kockelsberg	Bare fallow	8	8	8 m^2	2.0	P		718	1.1	S	R87
4	Marburg	Bare fallow	9	8	16 m^2	13.0	P	45/44/7	1230	6.4	S	J80

4	Marburg	Arable crops	9	8	16 m^2	24.0	P	45/44/7	1230	0.6	S	J80
4	Marburg	Bare fallow	9	8	16 m^2	2	P	45/44/7	1230	6.2	S	K56
4	Mittelgebirge	Bare fallow	8.0			3	P			2.9	S	P77
4	Olewig	Spring barley	8	8	8 m^2	3.0	P		465	0.0	S	R87
4	Olewig	Bare fallow	8	8	8 m^2	2.0	P		465	1.8	S	R87
4	Rauischholzh.	Bare fallow	8	8	16 m^2	9.0	P	22/48/30	418	1.6	S	J80
4	Rauischholzh.	Arable crops	8	8	16 m^2	26.0	P	22/48/30	418	0.1	S	J80
4	Schwäb. Alb	Bare fallow	6.0			5	P			13.9	S	D68
4	ZI1	Wheat	11.4	165	2145 m^2	1	W	8/74/7	1100	0.2	S	D86
5c	Erndtebrück	Arable crops	10.5	8	16 m^2	44.0	P	19/42/7	652	0.3	S	J80
5c	Erndtebrück	Bare fallow	10.5	8	16 m^2	12.0	P	19/42/7	652	0.7	S	J80
5c	Euchen	Bare fallow	7.5	10	15 m^2	0.9	P	4/75/21	569	22.1	S	B91
5c	Geisenheim	Vines	10–32	100		10	P	2 soils	625	151	S	E92
5c	Geisenheim	Vines + grass	10–32	100		5	P	45/23/12	625	0.001	S	E92
5c	Geisenheim	Vines	10–32	70		1	P	45/23/12	625	12.4	S	E92
5c	Geisenheim	Vines	10–32	30		1	P	45/23/12	625	3.1	S	E92
5c	Hochdahl	Bare fallow	8.0	10	15 m^2	0.6	P	63/30/7	842	42.1	S	B91
5c	Mertesdorf	Vines	38	8/16		168	P	31/19/11	602	0.2	S	R91
5c	Niederkasten.	Bare fallow	5.0	10	15 m^2	0.9	P	20/46/17	550	38.0	S	B91
5c	Odenwald	Forest		8	16 m^2	5	P	2 soils	780	0.003	S	V78
5c	Odenwald	Small grain	4–8	8	16 m^2	5	P	2 soils	780	0.19	S	V78
5c	Odenwald	Meadow		8	16 m^2	5	P	2 soils	780	0.19	S	V78
5c	Saalhausen	Bare fallow	18.5	10	15 m^2	0.5	P	18/39/13	780	1.3	S	B91
5c	Soest	Bare fallow	6.0	10	15 m^2	0.9	P	4/71/20		0.5	S	B91
5c	Tarforst	Spring barley	8	8	8 m^2	3.0	P		680	0.1	S	R87
5c	Tarforst	Bare fallow	8	8	8 m^2	2.0	P		680	0.5	S	R87
5c	Tarforst	Spring barley	8	8	8 m^2	3.0	P		680	0.1	S	R87
5c	Taunus	Rotation	8–13	8	16 m^2	5	P	6/77/16	650	5.6	S	V78
5c	Werden	Bare fallow	5.0	10	15 m^2	0.8	P	6/78/16		6.0	S	B91
5e		Bare fallow	9.5	8	8 m^2	20	P	4 soils	725	24.2	S	M88, A93
6		Bare fallow	9.5	8	8 m^2	70	P	14 soils	725	35.8	S	M88, A93
6	Au	Hops	5.9	65		34	T	24/49/23	750	55	S	S80
6	Au	Hops	2.4	70		45	T	27/55/10	750	15	S	S80

(*Continued*)

TABLE 1.18.A1 (*Continued*)

Landscape unit	Location	Land use	Slope (%)	Slope length (m)	Area	Plot years	Type of study	Soil texture Sa/Si/Cl (% bulk soil)	Rainfall (mm yr^{-1})	Mean erosion (t ha^{-1} yr^{-1})	Erosion type	References[a]
6	Au	Hops	4.6	50		45	T	28/50/14	750	24	S	S80
6	Geroldshausen	Hops	4.5	130		31	T	18/60/16	750	77	S	S80
6	Geroldshausen	Hops	6.2	210		22	T	56/28/9	750	205	S	S80
6	Mainburg	Hops	4.7	45		45	T	25/50/16	750	52	S	S80
6	Scheyern	Mixed. with mulch tillage	Av. 8.9	Av. 159	0.8–16 ha	60	W	Different soils	834	2.5	S	A03
6	Scheyern	Mixed. organic farming	Av. 11.4	Av. 111	1.6-11 ha	40	W	Different soils	834	0.2	S	A03
6	Scheyern	Small grain	20		2 ha	23	T	20/30/18	725	14	S	S02
6	Scheyern	Small grain	av. 10		81 ha	event	M			1049 t ha^{-1}	E	A98
7		Bare fallow	9.5	8	8 m^2	10	P	2 soils	725	15.6	S	M88, A93
7	Hohenpeißenb.	Arable crops	12	8	16 m^2	24	P	40/27/12	1304	0.3	S	J80
7	Hohenpeißenb.	Bare fallow	12	8	16 m^2	5	P	40/27/12	1304	2.5	S	J80
8	Brunnen	Pasture	36		1.8 ha	0.4	W		1065	0.48	S	F93
8	Grat	Pasture	34		301 m^2	0.4	W		1065	0.016	S	F93
8	Jenner	Pasture	45	40	182 m^2	0.4	P		1065	0.012	S	F93
8	Kaser	Pasture	36		2977 m^2	0.4	W		1065	0.34	S	F93
8	Königsbach	Pasture	36	40	186 m^2	0.4	P		1065	0.0006	S	F93
8	Königstal	Pasture	65	40	167 m^2	0.4	P		1065	0.001	S	F93
8	Wald	Forest	58		577 m^2	0.4	W		1065	0.0002	S	F93
8	Weg	Foot path	42		988 m^2	0.4	W		1065	4.80	S	F93
2	Gottesgabe	Arable crops	0	180 m	2.25 ha	2	W	88/6/6	475	20	W	F95
3	Ems-Hunte	Mixed landuse					W			<0.4	W	B03
3	Grönhiem	Bare	0	ca 300	7.55 ha	event	W	83/?/?		7.1 t ha^{-1}	W	G03
6	Mainburg	Hops	4.7	45		45	T	25/50/16	750	28	T	S80
6	Au	Hops	5.9	65		34	T	24/49/23	750	39	T	S80
6	Au	Hops	2.4	70		45	T	27/55/10	750	27	T	S80
6	Geroldshausen	Hops	4.5	130		31	T	18/60/16	750	42	T	S80
6	Geroldshausen	Hops	6.2	210		22	T	56/28/9	750	63	T	S80
6	Au	Hops	4.6	50		45	T	28/50/14	750	32	T	S80
6	Scheyern	Small grain	20		2 ha	23	T	20/30/18	725	21	T	S02

[a]A93, Auerswald (1993); A98, Auerswald (1998); A03, Auerswald *et al.* (2003a); B91, Botschek, (1991); B03, Böhner *et al.* (2003); D68, Dubber (1968); D86, Dikau (1986); E92, Emde (1992); F93, Felix and Johannes (1993); F95, Funk (1995); F98, Frielinghaus (1998); G89a, Goeck (1989); G89b, Goeck and Geisler (1989); G03, Goossens and Gross (2003); J80, Jung and Brechtel (1980); K56, Kuron *et al.* (1956), after Dikau (1986); M88, Martin (1988); P77, Preuss (1977); R87, Richter (1987); R91, Richter (1991); S80, Schwertmann and Schmidt (1980) (recalculated; this chapter); S02, Schimmack *et al.* (2002); V78, Voss (1978).

1.19

Switzerland

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1.19.1 INTRODUCTION

Early references to soil erosion in Switzerland mostly related to soil loss in vineyards (Peyer, 1958; Riva, 1973). However, soil erosion by water[1] was increasing from the 1950s onwards in Swiss agriculture, and was clearly visible in the areas directly affected. Only a few people, mostly from the scientific community, concerned themselves with the consequences, while others (in agronomy and policy) neglected the erosion hazard in the beginning. Since the mid-1970s, emerging off-site impacts, especially the eutrophication of various Swiss lakes, and the later Swiss National Research Program (NFP) 22 'Soil Use in Switzerland' (funded by the Swiss National Foundation, SNF) drew the attention of both media and policy groups to the damage and non-beneficial effects of this situation. Because of the importance of soil as a basis for the lives of humans, animals, plants and micro-organisms, and the irretrievable loss of quality, Swiss policy is increasingly taking more and more account of soil and water protection through a re-orientation of agricultural legislation.

An examination of soil erosion has to be undertaken against the background of Switzerland's heterogeneous landscapes in the first section. The predominant processes, soil formation rates and the reasons for increasing or decreasing soil erosion rates are the themes of the second section. Knowledge of on- and off-site damage and on- and off-farm costs (third and fourth sections) are the basis for policy decisions by legislators and also for soil conservation by the farmers. Both are discussed in the fifth section. Table 1.19.5 provides an overview of research done by Swiss federal institutes and universities.

[1]Soil erosion caused by wind-driven forces is a minor problem in Swiss agriculture. Because of the small contribution of wind erosion to the total, each part in this country review relates to water erosion.

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TABLE 1.19.1 Rainfall characteristics, soil properties and land use in the geographical regions

Geographical region	Precipitation (mm) *R*-factor ($N\,h^{-1}$)	Predominant soils; *K*-factor ($kg\,h\,N^{-1}\,m^{-2}$)	Land use (the first land use type is slightly predominant)	References: with focus on the geographical region
High Rhine Valley	1000–1050 65–90	Luvisol; calcaric Cambisol; 0.45–>0.6	Mainly arable land	Schaub (1989); Hebel (2003)
Jura	850–>2000 65–>100	(Eutric, vertic and stagnic) Cambisol, rendzic Leptosol, (stagnic) Luvisol; 0.1–>0.4, mostly 0.2–0.3	Grassland and arable land	Vavruch (1988); Prasuhn (1991); Hebel (2003)
Swiss Plateau	900–1250 70–100	(Eutric and stagnic) Cambisol,(stagnic) Luvisol, Gleysol; 0.2–0.55	Arable land and grassland	Crole-Rees (1990)
Pre-alpine Swiss Plateau	1000–2000 100–>140	Cambisol, Gleysol, Podzol; 0.2–0.45	Mainly grassland	Rohrer (1985)
Alps	⟨600–⟩4000 Highly variable	Cambisol, Podszol, (rendzic and dystric) Leptosol, Fluvisol, Phaeozem	Grassland	
South of the Alps	1,300–>3,000 Up to > 700	Cambisol, Podzol (dystric) Leptosol; 0.1–0.25[a]	Mainly grassland	Marxer (2003)
References: national scale	Schaub, 1989; Thöni, 1990	Schaub, 1998; Neyroud, Müller, 1990	BFS 2003	Mosimann *et al.* (1990, 1991); Schaub and Prasuhn (1998)

[a]Data from four test sites (Marxer, 2003).

1.19.2 GEOGRAPHICAL REGIONS AND SOIL EROSION

The Swiss landscape can be roughly divided into three geographical regions (Figure 1.19.1):

- Jura: karst mountains (limestone, clay, marl);
- Swiss Plateau: Pleistocene river terraces, morainic deposits (gravel, sands, loesses) and molasse (sandstones, marls, conglomerates); and
- Alps: molasse, limestones, crystalline, slate.

Soil erosion risk is based on an adaptation of USLE made by Schaub and Prasuhn (1998) (also published in Schaub, 1998). The calculation for municipal areas is based on officially available data. Areas without shading are not classified (arable land less than 5% of the total agricultural area) or where soil erosion risk is less than $1\,t\,ha^{-1}\,yr^{-1}$. The soil erosion susceptibility layer is an overlay of the erosion risk with the actual percentage of arable land for each municipality.

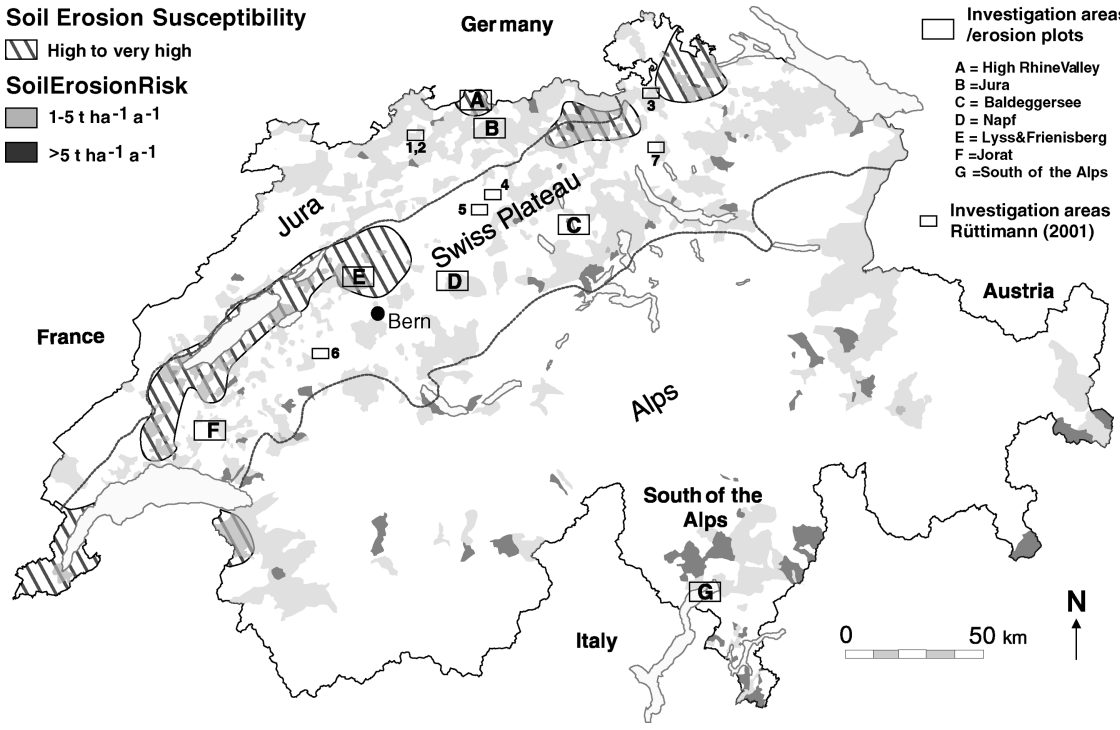

Figure 1.19.1 Geographical regions and erosion risk in Switzerland. (Source of erosion risk, Schaub, 1998, modified; soil erosion susceptibility, Schaub and Prasuhn, 1998; very generalized, Leser *et al*., 2002)

As far as soil erosion processes and rates are concerned, the study also has to cover the Pre-Alps as the southern part of the Swiss Plateau, the region of Mediterranean influence south of the principal Alpine chain and a small area in the northern region, the High Rhine Valley.

Rainfall occurs all through the year, with spatial distribution influenced by the relief. In general, rainfall erosivity[2] is moderate, but increases with the amount of rainfall. Around 80% of annual erosivity is caused by rainfall during April and September, with the highest erosivity normally occurring from June to August (Mosimann *et al.*, 1990).

The *High Rhine Valley* (about 20 km east of Basel), which occupies about 1% of Swiss arable land, consists mainly of Wurm-aged lower terrace with gravel and Riss-aged higher terrace with loess cover. Owing to the relatively high erodibility of these soils (Table 1.19.1), high erosion rates can be observed in this area (Schmidt, 1979; Schaub, 1989, 1998; Unterseher, 1997; Hebel, 2003). A high clay content of soils and small fields result in a moderate erosion risk in the *Jura*. Up to the 1980s, maize was still a common crop here, resulting in higher erosion rates on these fields. The predominant type of land use is actually grassland. Nevertheless, soil erosion occurs on fields owing to their location on relatively steep slopes.

In general, erosivity is lower in regions to the leeward of the Jura, and increases again in the pre-alpine parts of the *Swiss Plateau*. In some areas, especially where glacial till and fine sediments (e.g. loess) are present, the *K*-factor rises to 0.55 (Mosimann *et al*., 1990; Neyroud and Müller, 1990). In the rounded relief in the eastern Swiss Plateau, soil erosion is encouraged by slope shape and sandy-loamy soil textures. Compared with the

[2]*R*-factor of Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978).

Jura and the Pre-Alps, larger field sizes with lower slope gradients are cultivated in the Swiss Plateau. Nevertheless, land consolidation, in combination with moderate to locally high soil erodibility and intensive crop production, contribute to relatively high soil erosion rates. About 32% of arable land in Switzerland is cultivated with root crops (sugar beet, potatoes) and maize, including most of the area of the Swiss Plateau. These crops increase the soil erosion risk in general. In addition, the root crops also result in soil loss through the soil aggregates attached to and harvested with the roots.

In the *pre-alpine Swiss Plateau* (Pre-Alps), rainfall and erosivity are higher owing to the nature of the relief. In this hilly region, the erosion risk is normally low to moderate, because of the predominance of grassland and the well-adapted crop rotation on arable land. Local rates may be higher owing to heterogeneous soil properties and more susceptible circumstances, e.g. vineyards (Rohrer, 1985; Schaub and Prasuhn, 1998).

The highest erosivity due to rainfall in Switzerland occurs naturally in the *Alps*. Rainfall erosivity for this region shows great variations due to relief, and is many times higher than in the Jura and the Swiss Plateau. Land use has become well adapted to the steep slopes, high precipitation and erosivity. Arable land is mostly situated in flat alpine valleys, where wind erosion may affect alluvial deposits and marshy and sandy soils (Mosimann *et al.*, 1990). In grassland, damage is mostly a result of related processes caused by management, soil creep, earth flow and other processes of mass movement. Soil erosion itself seems to be a minor problem. Nevertheless, in the region *south of the Alps*, with a Mediterranean-influenced climate, soil erosion risk is temporarily higher when the vegetation cover decreases locally owing to forest fires (Marxer, 2003).

Soil erosion increased from the 1950s to the 1990s owing to the following circumstances (Mosimann *et al.*, 1990, 1991; Mosimann, 2003; Ogermann *et al.*, 2003, statistical data source: BFS, 1989, 2003):

- An increase in the area of arable land between 1965 and 1990 from 248 901 to 312 606 ha (+26%), particularly on sloping sites, due to the spread of settlements in flat and valuable areas.
- Land consolidation in the 1960s resulted in fields of 4–8 ha in size (mostly on slopes) and a reduction in natural retention elements such as hedges.
- Additional land levelling for more homogeneous fields.
- Extension of silage maize cultivation from 5 226 ha in 1965 to 38 797 ha in 1990 (+742%). Maize and silage maize together increased from 17 555 ha (1975) to 66 178 ha (1990) in combination with increasing cultivation of sugar beet (+65%), with higher erosion rates occurring during the more erosive season.
- Reduction of temporary grassland (−28%) in crop rotation during the same period.
- Deterioration of soil structure due to intensification of mechanical soil management and pressure, and partly to a low organic matter content.
- Vines are cultivated in nearly every region, mostly on steep to very steep slopes, owing to microclimatic conditions. Despite the soil cover, high rates of soil loss are the result (up to 15 t ha^{-1} during heavy rainfall).

Since the 1980s, soil erosion has been decreasing in the Jura owing to a change to more temporary grassland in crop rotation as well as a complete change to grassland.

1.19.3 EROSION PROCESSES AND RATES IN SWISS AGRICULTURE

Moderate erosivity of rainfall and erodibility of soils and also relatively small fields in Switzerland do not lead to very spectacular on-site damage (Tables 1.19.2 and 1.19.3). Nevertheless, with respect to the sustainable use of soil resources and protection of surface water, soil erosion in Switzerland is probably too high in some regions and may exceed tolerable on- and off-site rates. Soil formation processes are generally very slow. Mosimann *et al.* (1991) assumed the following rates of soil formation for Switzerland (all in mm yr^{-1}): High Rhine Valley (or other loess-covered areas), 0.15–0.2; Jura, 0.02–0.05 (limestone) and 0.05–0.1 (marl); and

TABLE 1.19.2 Soil erosion rates and processes in different geographical regions of Switzerland (surveyed with different methods)

Area	Period	*R*-factor ($N\,h^{-1}$)	*K*-factor ($kg\,h\,N^{-1}\,m^{-2}$)	Soil loss ($t\,ha^{-1}\,yr^{-1}$)	Sheet erosion (%)	Rill erosion (%)	Other forms (%)	Source
High Rhine Valley	1974–1999	90	0.45–>0.6	0.3–0.5	65/40	10/10	25/50	Schaub (1989, 1998)
Jura	Since 1978	105–115	0.1–>0.4, mostly 0.2–0.3	0.6–1.9	5	10	85	Prasuhn, (1991); Schaub, (1998)
Swiss Plateau (Lyss)	1987–1989	80–90	0.2–0.55	0.3–1.0	50	20	30	Mosimann *et al.* (1990)
Swiss Plateau (Frienisberg)	Since 1997	80–90	0.33	0.91	34	66	—[f]	Prasuhn and Grünig (2001)
Pre-Alps (Napf)	1980–1982	140	0.2–0.45	0.3–0.5	3	74	23	Rohrer (1985); Schaub (1998)
South of the Alps[a]	1996–1997	645-735[b]	0.1–0.21[c]	0.6–19.0[d] 0.2–6.5[e]	—[f]	—[f]	—[f]	Marxer (2003)

[a]Soil erosioin rates in chestnut forests after wild fires.
[b]Two meteorological stations: 1986-1999.
[c]Data from 3 test sites.
[d]1st post-fire year (from spring onward).
[e]2nd post-fire year.
[f]No survey was made by authors.

Swiss Plateau, 0.1 or slightly more. Investigations of the rates of soil formation in the Jura show serious difficulties in determining these rates in non-homogeneous substrates, e.g. Opalinus clay (Schwer, 1994). In fact, the above-mentioned rates are scientifically insecure; according to Bork (1988), there is probably no new soil formation on arable land under current Central European climatic conditions and management. Therefore, even small rates of soil erosion may be too high for on-site areas, although they may be tolerable for off-site impacts.

A country-wide assessment of the state of soil erosion damage and risk on arable land was one focus of NFP 22 from 1987 to 1990 (Mosimann *et al.*, 1990, 1991; Häberli *et al.*, 1991). An estimate of the countrywide susceptibility to soil erosion based on Allgemeine Bodenabtragsgleichung (ABAG) (Schwertmann *et al.*, 1987) was presented by Schaub and Prasuhn (1998) with a digital elevation model, official data and statistics for municipalities. This national overview indicates some crucial areas in Switzerland (Figure 1.19.1) where soil conservation measures are recommended. But a resolution of approximately 6 km^2 does not allow statements on soil erosion susceptibility and rates on an individual field scale.

Some 30 % of the arable land on the loess-covered area in the *High Rhine Valley* exceeds tolerable values. An average soil loss of about 5 $t\,ha^{-1}\,yr^{-1}$ (on some fields up to 95 $t\,ha^{-1}\,yr^{-}$) makes this area one of those most affected in Switzerland (Schaub, 1989). Because of moderate steepness and length of slopes and soil properties, the High Rhine Valley is characterized by frequent interrill erosion.

In the *Jura*, the dominant soil erosion causes are Thalweg erosion, exfiltrating interflow (together about 60 %) and snowmelt erosion (Vavruch, 1988; Seiberth 2001); sheet erosion is minimal, and rill erosion occurs predominantly in preferential superficial flowpaths. Thalwegs may show wide but shallow rills. In clay soils it is mostly soil aggregates that are transported. These accumulate with decreasing transport capacity of surface runoff in the foot slope area (Schaub, 1998).

TABLE 1.19.3 Long-term measurements of soil loss rates and surface runoff on the test plots (T1 etc.) in the High Rhine Valley, the Jura, the Pre-Alps and south of the Alps (test plots are all bare soil, except for those south of the Alps, which are in chestnut plantations after forest fire)

	High Rhine Valley[a]		Jura[a]		Pre-Alps (Napf)[a]		South of the Alps[b] (soil erosion due to forest fires)		
	T1	T2	T30	T50[c]	T300	T350	Contra	Ronco S. Ascona	S. Antonino
Period of measurements	1975–1999	1975–1984	1978–1995	since 1983	1980–1982	1980–1982	04/96–12/97	03/97–11/98	03/98–12/99
Number of parcels	3	3	3	1	3	2	2	6	6
Size of plot(m)	1 × 10	1 × 10	2 × 10 (2) 1 × 10 (1)	3 × 20	1 × 10	1 × 10	3 × 10	3 × 10	3 × 10
Slope (%)	14	13	17	21	31	29	72, 95	48–68	46–53
Erosivity (*R*-factor) ($N\,h^{-1}$)	90	90	115	105	140	140	555[d] 932[e]	353[e] 523[e]	663[d]1,078[d]
Erodibility (*K*-factor) ($kg\,h\,N^{-1}\,m^{-2}$)	0.26	0.52	0.22	0.20	0.24	0.33	0.14	0.21	0.22
Average annual soil loss ($t\,h\,a^{-1}$)	13.7	21.5	13.4	21.6	8.6	22.3	19.0[d] 6.5[e]	0.6[d] 0.2[e]	0.9[d] 0.6[e]
Average annual surface runoff ($l\,m^{-1}$)	15.8	27.1	7.6	11.4	36.0	55.0	61.1[d] 50.9[e]	106.5[d] 82.6[e]	43.83[d] 39.63[e]
Sediment concentration ($g\,l^{-1}$)	86.9	79.2	176.4	189.0	23.8	40.5	30.5[d]14.2[e]	0.6[d] 0.2[e]	2.0[d] 1.2[e]
Runoff (% of total precipitation)	1.9	2.9	0.7	1.1	2.5	3.7	—	—	—
Runoff (% of erosive precipitation)	11.9	23.4	9.5	7.5	11.1	14.0	—	—	—

[a]Sources (summarized by Schaub, 1998): Rohrer (1985); Prasuhn (1991); Schaub (1989); Schaub and Prasuhn (1993); Schmidt (1979); Seiler (1983) Unterseher (1997); Vavruch (1988).
[b]Source: Marxer (2003).
[c]The test plot T50 is still in use; the presented data are calculated up to 1998.
[d]1st post-fire year (from spring onward).
[e]2nd post-fire year.

In the *Swiss Plateau*, 10–40% of arable land is threatened regularly by soil erosion (Mosimann *et al.*, 1991). Sheet erosion there is three to five times more widely distributed than concentrated runoff. However, soil loss rates through linear erosion are frequently higher than sheet erosion during an event.

According to Mosimann *et al.* (1990, 1991), the most important erosion risk factors on the Swiss Plateau are:

- thalwegs, inflow of upslope water, high proportion of root crops and maize in crop rotation;
- large fields and slope lengths, high mechanical pressure on the soils, slope steepness and management in the slope direction; and
- soil properties (texture, organic matter content and permeability).

In the event of storms or abundant rainfall in combination with snowmelt processes, soil losses up to 40 t ha^{-1} may occur. During the NFP campaign, a very local storm (with hail) led to soil loss rates up to 90 t ha^{-1}. In some fields (concentrated), overland flow truncated the soil profile by about 4 cm and thalwegs were eroded up to 15 m in width (Mosimann *et al.,* 1990, 1991). On a 22-ha investigation area, 74% of 2370 t (about 108 t ha^{-1}) of eroded material was entering the surface water (Rohr, 1994).[3]

On the relatively steep slopes of the *Pre-Alps*, rill erosion is mainly caused by management on the slope and the cultivation of potatoes (Rohrer, 1985; Schaub, 1998).

Providoli *et al.* (2002) studied the rates of splash erosion on soils after a fire in a chestnut plantation by comparison with a direct clear-cut and a non-intervention option. Absolute splash erosion rates obviously vary over different soil types. Within one soil type, neither the aggregate stability nor the development of soil cover was significantly different between the two management options.

1.19.4 ON- AND OFF-SITE DAMAGE AND COSTS

Soil erosion damage and costs are often low at farm level in the short term. On the other hand, the damage and costs for the community as a whole are considered to be higher on account of non-point-source pollution, eutrophication, sedimentation and more frequent high water events, owing to reduced retention in first-order catchment areas.

Mosimann *et al.* (1991) presumed that 10–20% of the soil erosion material will enter into river systems in the Swiss Plateau, with 40% remaining on the field. They estimate about 0.4 t ha^{-1} yr^{-1} soil loss to surface water for Switzerland. According to Prasuhn (1991), about 20% of eroded soil is fed into rivers in the Jura.

Wilke and Schaub (1996) investigated the phosphorus (P) enrichment ratio for different regions of Switzerland. The analysis of 223 samples (both eroded and source material) shows a general enrichment of P in eroded soils. Although there is an obvious enrichment compared with the source material, Wilke and Schaub did not find a correlation between soil loss rate and enrichment ratio. For better protection of off-site areas, especially surface water, they suggested using not the average enrichment ratio (1.35) for calculations, but rather 1.86 (the highest value) for Switzerland. According to calculations by Braun *et al.* (1994), the annual P loss from Swiss arable land into surface water is about 1100 t, which is contributing to eutrophication, mainly of lakes. Additional P loss is caused through runoff from grassland (ca 570 t P yr^{-1}) and leaching (ca 230 t P yr^{-1}).

[3]A mapping method for erosion damage was developed by the Soil Erosion Research Group Basel (FBB) (Rohr *et al.,* 1990) and was integrated in the later DVWK method (DVWK, 1996). Actual mapping of erosion damage is carried out in small investigation areas in the Swiss Plateau by the Swiss Federal Research Station for Agroecology and Agriculture (FAL) (Prasuhn and Grünig, 2001; Prasuhn and Weisskopf, 2003) and in the Jura by the FBB (Leser *et al.*, 2002).

Despite the already mentioned retention elements in the Jura, the concentrations of river sediment and accompanying nutrients, etc., are high in the Jura during storm events, owing to hydrological shortcuts, e.g. conduit inlets between roads and fields and drainage inlets on the edge of fields in broad superficial flowpaths (thalwegs). Similar hydrological shortcuts are responsible for high soil and P inputs in surface waters on the Swiss Plateau. Prasuhn and Grünig (2001) measured and calculated soil erosion and the loss from arable fields to surface water through mapping and model calculations in five test areas of Frienisberg (part of the former NFP campaign, Figure 1.19.1). The aim of the ongoing study is the evaluation of the ecological measures (compare with Section 1.19.5). The results show a high artificial connectivity of arable land and surface waters:

- 65% of arable land is directly or indirectly linked to surface waters;
- 18% (90 m^3) of eroded soil entered the water;
- 90% of this 90 m^3 is indirectly due to street and path drains;
- Calculations using an average P content of arable soils of 960 mg kg^{-1} and the above-mentioned enrichment ratio of 1.86 give an average annual P input into surface waters of 0.3 kg P ha^{-1} of arable land.

Quantification of the costs of soil erosion for the on- and off-site areas depends mostly on the observed time-scale, which one can assume is difficult to calculate. Schmid *et al.* (1998) indicated that on-farm costs are of only minor significance for the optimal long-term behaviour of farmers in Switzerland. The off-site benefits of the reduction of non-point-source pollution from agriculture, according to Keusch (2001), are ''significant and are obviously able to exceed on-site benefits''. An example of off-site costs is eutrophication of the Baldeggersee lake, where water quality will be improved with treatment in the form of artificial oxygen which is put into the lake during summer. Keusch (2001) used a combined modelling approach, which includes the biophysical simulation model EPIC and an economic optimization model to analyse soil erosion and P loss in particular. According to Keusch, the on-farm costs of soil erosion (in a mid-term view) for the first year are between 6.88 and 27.44 Swiss francs[4] (SFr) per millimetre of eroded soil material per hectare, depending on the soil depth. Keusch further illustrates the costs to the farmers of avoiding P loss in different scenarios: for a limit of 0.3 kg ha^{-1} costs of at least 500 SFr will be generated per hectare and for a 0.1 kg ha^{-1} limit 1200 SFr.

1.19.5 SOIL CONSERVATION AND POLICY

National authorities are responsible for environmental legislation, economic incentives (direct payments), research and links of national and multinational programmes. One of these international programmes is to establish agri-environmental indicators (e.g. indicator 13: soil erosion risk), based on models of the Organization for Economic Cooperation and Development (OECD) and the European Commission (FAL 2003).

On top of that, various cantonal soil conservation initiatives are carried out, for example:

- Baselland: development of the 'Key for the assessment of soil erosion' (Mosimann and Rüttimann, 1995) and monitoring of erosion risk and soil conservation due to management (Mosimann, 2003);
- Bern: investigation of no-till methods and economic incentives (Vökt, 2001);
- Solothurn: development of a computer program to estimate soil erosion risk, based on the 'Key for the assessment of soil erosion', and soil erosion risk map for arable land;
- Vaud: thalweg erosion;

[4]Approximately 1.5 SFr = €1.

- Various cantons: modification of the 'Key for the assessment of soil erosion' (e.g. Mosimann and Rüttimann, 1996, 1999) and distribution of leaflets about soil erosion.

Studies of the reduction of soil erosion rates due to soil management (no-till, mulch seeding, etc.) are very limited in Switzerland, although there has been extended research on such technologies, mostly in the context of crop yield (e.g. Reinhard *et al.*, 2001).

A study of soil erosion rates and herbicide discharge under different systems of soil management was carried out by Rüttimann (2001). Mulch seeding of silage maize with winter hardy and non-winter hardy catch crops results in a significant reduction of runoff and soil erosion (Table 1.19.4). The author suggests that non-winter hardy catch crops are preferable to winter hardy crops, besides giving a higher yield. Minimizing runoff by mulch seeding can reduce herbicide loss by 80% on average. Rüttimann also disproved ''the fear that higher concentration of atrazine with minimum tillage would lead to a herbicide discharge comparable to conventional tillage despite reduced runoff and soil erosion''. Taking into account the different soils tested in the study, it is probable that pesticide loss is higher. For this reason, Rüttimann recommends in every case a combination of conservation tillage and additional action by farmers, e.g. minimization of herbicide use, optimization of application timing, reduction of point source loss from farms and creation of additional surfaces for water retention in rough, intensively cultivated watersheds.

For the sites Teufen and Buckten I, the maize period was only measured from the 'three-leaf-stage' onward until harvesting (three repetitions at each test location). A1 'plough' conventional tillage with uncovered soil during the catch crop period; inspection system; B1 'non-winter hardy catch crop' (white mustard or phacelia); B2, 'winter hardy catch crop' (winter rye); C1 minimum (broad) tillage 'rototiller'; C2, minimum (band) tillage 'band-rotary-hoe'.

Conservation tillage is increasing on the Swiss Plateau owing to various federal and cantonal programmes. Soil management was also investigated by Prasuhn and Grünig (2001). About 3% of arable land in the region of Frienisburg is cultivated by no-till methods, 4% by sowing with rotary band cultivators and 7% by mulch seeding. Today, 67% of arable land is still ploughed and cultivated intensively, although a decrease is obvious (in the late 1980s some 95% was ploughed). During the period 1987–89, no conservation tillage was practised in this region, so those kinds of soil management represent a development started in the 1990s. Furthermore, a decrease of bare fallow during winter from 22 to 5% has been observed in addition to a reduction of about 5% in arable land.

Prasuhn *et al.* (1997) give most priority in reducing P loss due to soil erosion on arable land in the Swiss Plateau to conservation tillage and to suitable crop rotation. For the central and eastern Swiss Plateau, soil erosion is often governed by slope form and slope angle. Erosion protection for those areas includes changing the direction of tillage, segmentation of slopes, or seeding of buffer strips (Schaub and Prasuhn, 1998).

Studies of the long-term effects of biodynamic (D), organic (O) and conventional (K) land-use management are carried out in north-west Switzerland in a long-term field trial, better known as the DOK trial, by the Research Institute of Organic Agriculture (FIBL). Within this study, experiments to determine the effects of this alternative farming system on earthworm population and soil erodibility were carried out by Siegrist *et al.* (1998). The aggregate stability and the tendency for soil sealing were highest on biological plots. Despite higher aggregate stability and 30–140% higher infiltration capacity, splash erosion was not reduced on biological plots, compared with conventional ones. Owing to higher infiltration rates, transport capacity was lower on biological plots.

The NFP 22 results relate to the context of the reorientation of Swiss agricultural policy in 1993, focused on a more ecologically orientated agriculture. This aim is to be achieved through direct payments for the implementation of ecological measures (Ökomassnahmen) by farmers. These ecological measures should generally reduce matter and nutrient loss from diffuse sources of agriculture to surface water. For example, the P input has to be reduced by about 50% by 2005 in comparison with the period before the inauguration of the

TABLE 1.19.4 Effects on soil erosion of various sowing methods for silage maize

Sites (code of Figure 1.19.1)	Investigation period	Size (ha)/ slope (%)	Precipitation (mm)/*R*-factor	*K*-factor	Precipitation[a] (mm)/average runoff rate (%)	Soil loss ($t\ ha^{-1}\ yr^{-1}$) during maize period				
						A1	B1	B2	C1	C2
Buckten I (1)	1989/1990	0.87/20	986/80	0.23	383/3.02	6.10	0.39	0.00	1.38	0.00
Buckten II (2)	1989/1990	1.12/18	986/80	0.20	303/0.00	0.00	0.00	0.00	0.00	0.00
Teufen (3)	1989/1990	1.14/15	951/77	0.32	334/0.77	1.82	0.36	0.00	0.00	0.15
Obermuhen (4)	1990/1991	1.29/15	1130/92	0.30	346/2.40	5.35	0.36	0.29	0.13	0.00
Hirschthal (5)	1991/1992	0.76/14	1130/92	0.32	382/3.40	16.34	0.52	0.20	0.79	0.11
Wiler (6)	1991/1992	0.71/14	1191/97	0.26	383/3.55	8.01	0.00	0.00	0.37	0.04
Rümlang (7)	1990/1991	0.89/9	1013/82	0.36	377/3.29	3.34	0.92	0.70	0.36	0.65

[a]During maize period.

Source: Rüttimann (2001) (modified).

programme. Since 1999, the condition for receiving direct payments is proof of the fulfilment of the ecological requirements [Ökologischer Leistungsnachweis (ÖLN)]. In the case of the on- and off-site consequences of soil erosion, the ÖLN conditions of particular interest are as follows (Prasuhn and Grünig, 2001; Prasuhn and Weisskopf, 2003):

- a well-balanced fertilizer budget;
- a suitable role for ecological compensation areas;
- well-adapted crop rotation;
- proper soil protection; and
- careful selection and application of pesticides.

The recommendations of the NFP 22 also constituted a basis for soil-protection legislation. Assuming that soil formation is a very slow process under current land-use and climatic conditions, even low erosion rates harm the long-term fertility of on-site areas and the quality of off-site systems. This was taken into account by the Swiss Council of Ministers in its revision of the Swiss Environmental Protection Regulation and the Ordinance on Soil Protection [Verordnung über Belastungen des Bodens (VBBo); Schweizer Bundesrat, 1998]. The VBBo establishes threshold values for average soil loss on arable land for long-term soil fertility. The maximum permanent soil loss due to erosion depends on soil depth, and may not exceed $4\,t\,ha^{-1}yr^{-1}$ (>70 cm soil depth) or $2\,t\,ha^{-1}yr^{-1}$ (<70 cm).

These threshold values for soil loss rates should be controlled by the cantons through the USLE-based 'Key for the assessment of erosion risk'. This method also allows a rough estimate of the erosion risk for farmers and advisory services to be obtained on the basis of fixed standards (Mosimann and Rüttimann, 1995, 1996, 1999; Schaub and Prasuhn, 1998; Charollais and Schaub, 1999).

Sustainable soil utilization in general, and the reduction of soil erosion in particular, are also integrated into the Swiss environmental protection regulations [Umweltschutzgesetz (USG)] and the revised Swiss agricultural legislation [Landwirtschaftsgesetz (LWG)]. Integrated production (IP), buffer strips (3 m wide) are necessary in the riparian zone and are regulated by the LWG. The extensive use of these zones should reduce diffuse nutrient input from arable land to surface waters.

1.19.6 CONCLUSION

Switzerland is the first European country which has established threshold values for average soil loss on arable land. Owing to its soil-protection policy, federal programmes (e.g. ecological measures) and cantonal initiatives (Table 1.19.5), Switzerland possess various types of soil-conservation policy strategies for long-term soil fertility. Even with relatively moderate erosion risk, soil erosion in Switzerland causes distinct damage and costs in both on- and off-site areas. Soil erosion on arable land is not the only process contributing to off-site problems, but in fact is one of the major processes in some regions. Through reorientation of Swiss agricultural policy, soil erosion should be minimized by conservation tillage, buffer strips, adapted crop rotation, etc. Nevertheless, the projected 50% reduction of P input from fields to surface water will probably not be reached by 2005 (Prasuhn and Grünig, 2001).

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TABLE 1.19.5 Federal Institutes and Universities working on soil erosion in Switzerland

Recent projects	Swiss Federal Research Station for Agroecology and Agriculture (FAL) http://www.reckenholz.ch
Geographical Institute Basel, Physical Geography and Landscape Ecology http://www.physiogeo.unibas.ch	Swiss Federal Institute of Technology (ETH) Zurich, Institute of Agricultural Economics (IAW) http://www.iaw.agrl.ethz.ch/
Former or related projects	Swiss Federal Institute of Technology (EPFL) Lausanne, Institut des Sciences et Technologies de l'Environnement http://iste.epfl.ch/
Swiss Federal Institute of Technology (ETH) Zurich, Institute of Plant Sciences (Plant Nutrition Group) http://www.pe.ipw.agrl.ethz.ch Institute of Terrestrial Ecology (ITÖ) http://www.ito.umnw.ethz.ch	Swiss Agency for the Environment, Forests and Landscape (BUWAL), including cantonal authorities for the environment http://www.umwelt-schweiz.ch/buwal/de/
Swiss Federal Institute for Forest, Snow and Landscape Research (WSL) http://www.wsl.ch	Swiss Federal Research Station for Agricultural Economics and Engineering (FAT) http://www.sar.admin.ch/fat/
Swiss Federal Research Station for Plant Production (RAC) http://www.sar.admin.ch/rac	Research Institute of Organic Agriculture (FIBL) http://www.fibl.org/
Swiss College of Agriculture (SHL) http://www.shl.bfh.ch/	Schweizerische Gesellschaft für bodenschonende Landwirtschaft (Swiss no-till) http://www.no-till.ch

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Owing to the existence of an extensive literature on soil erosion in Switzerland, the reference list has had to be summarized as much as possible. An extended catalogue of Swiss soil erosion literature can be requested at http://www.physiogeo.unibas.ch/ (publications) or from the authors.

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1.20

Italy

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1.20.1 INTRODUCTION

Soil erosion is a complex process that depends on climate, relief, soil and vegetation. Effects of human activities on the landscape have been present in Italy for the last 3000 years. The Italian landscape has been modified by land use (e.g. terracing) and much of Italy exhibits a clear human imprint. At present, the great mechanical power available to any farmer has made human interaction with the environment extremely efficient. Removal of terraces and the transformation of hillslopes into a good approximation of inclined planes are two aspects of the changes occurring to the landscape. The rate at which such change is taking place is impressive, and previously unmatched. As a consequence, characteristics and qualities of soils are severely affected, mainly through processes that can be loosely described with the term 'soil loss', which encompasses water erosion, badland erosion, mass movements, mechanical erosion (i.e. tillage, land levelling, root crop harvesting) and wind erosion. Since wind erosion is present in only a few, limited, areas in Italy, it will be omitted from this chapter.

1.20.2 WATER EROSION

Water erosion represents one of the most important and widespread causes of soil degradation in Italy. About 77 % of Italy is at risk of accelerated water erosion (Gazzolo and Bassi, 1961) owing to the steep slopes, the

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intense mechanical cultivation, the increased size of fields, the elimination of conservation measures such as terraces and waterways (Chisci, 1986) and the exposure of more erodible subsoil materials.

Soil erosion peaked approximately during 1930s–1940s, when considerable parts of Italy were turned into arable land. In the 1950s and 1960s, industrialization led to the abandonment of agricultural land, which resulted in a reduction in sediment yield (Panicucci, 1971; Panicucci and Maletta, 1982; Bazzoffi *et al*., 1986), contributing to an increased rate of coastal erosion (CNR, 1982). Since the 1970s, part of the previously abandoned terrain was turned into arable land again, which, in many places, meant renewed erosion. Even if today soil erosion affects less of the territory than it did 60 years ago, the combined effects of specialized crop management and the techniques used for implementing land-use changes have led to an increase in soil erosion rates.

Rough estimates of the potential risk of soil erosion due to rill and inter-rill processes in Italy have been made by Van der Knijff *et al*. (2002) and Grimm *et al*. (2002). Such estimates are based on the application of the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978), and were made using the available data sets, with adaptations and simplifications that resulted in maps of difficult interpretation. A different approach was later attempted by Van Rompaey *et al*. (2003), which used measurements of erosion in 21 catchments to control the model outputs.

The only integrated attempt at assessing soil loss in Italy was carried out in the framework of a 5-year national project which ended in 1982 (CNR – Finalized Project 'Soil Conservation'; CNR, 1982). If one excludes CORINE (1992) and, more recently, the Pan European Soil Erosion Risk Assessment initiative (PESERA, http://pesera.jrc.it), both funded by the European Commission, no other systematic efforts have been made to determine soil erosion rates in Italy. This means that surveys of soil erosion at the national scale do not exist, and that the few data available are spatially clustered in a limited number of sites, as a consequence of local research interests or problems. Recently, the situation has improved, largely as a result of the activities of the river Basin Authorities, instituted by the Soil Protection Act (183/1989) with the aim of planning and managing in an integrated way the catchments of the Italian rivers.

1.20.2.1 USLE Studies: Some Results

Studies to adapt the USLE to Italian conditions have been conducted over periods that are usually too short to attain the conditions required for a realistic comparison (i.e. about 20 years; Wischmeier and Smith, 1978) and took into account only some of the factors of the equation. In the following we summarize the main results obtained in Italy.

1.20.2.2 Rain Erosivity

The only attempt at testing whether the equation linking rainfall characteristics to rain kinetic energy (E) in Italy was made by Zanchi and Torri (1980), who found that a way of considering the marked seasonal diversity of raindrop size distribution during a typical year was by adding air temperature (T) to rain intensity (I). The equation is

$$E(\mathrm{J\,mm^{-1}}) = 1.86 - 11.03\log I + 6.07\log T \qquad (1.20.1)$$

No attempts were ever made to question the validity of the erosivity index, EI_{30}, and particularly to determine whether I_{30}, I_{10} or I_{15} is the best parameter for estimating soil loss. Attempts have been made at using the erosivity index EI_{30} as designed by Wischmeier and Smith (1978), using either the standard technique or some simplified versions of it. In this respect, D'Asaro and Santoro (1983) published a rain erosivity map of Sicily. More recently, Van der Knijff *et al*. (2002) and Gabriels (2002) calculated rain

erosivity for Italy using two different approaches. Van der Knijff *et al*. (2002) based their maps on a personal communication by Zanchi, which suggested that erosivity in Tuscany (central Italy) is proportional to the cumulative yearly rainfall ($EI_{30} = aP,\ a \in [1.1; 1.5]$ in (MJ mm)(ha h a)$^{-1}$ and P in mm). Gabriels (2002) used the modified Fournier index (Arnoldus, 1980), defined as follows:

$$MFI = \sum_{1}^{12} \frac{P_j^2}{P} \tag{1.20.2}$$

where P is the average annual rainfall (mm) and P_j is the average rainfall of the jth month.

So far, neither of the two approaches has been compared with experimental data. We have collected published data on erosivity determined as suggested by Wischmeier and Smith (1978). The data include the work by Zanchi (1976) and Bazzoffi and Pellegrini (1992) for Tuscany, by Linsalata *et al*. (1983) and De Franchi *et al*. (1983) for Lucania, by Calzolari *et al*. (2001) for Emilia Romagna and Tuscany and by D'Asaro and Santoro (1983) for Sicily. The complete data set was further expanded using the following relationship (Calzolari *et al*., 2001):

$$EI_{30} = 0.11 \sum P_j^{1.85} \tag{1.20.3}$$

where P_j is the rainstorm amount (mm) and the sum is extended to all the rain events in a year. This equation, which is similar to that developed by Richardson *et al*. (1983) and applied in Sicily by Bagarello and D'Asaro (1994), was compared successfully with appropriately calculated data on rain erosivity by Calzolari *et al*. (2001).

Figure 1.20.1a shows the relationship between the annual precipitation and rain erosivity in Tuscany, Abruzzi and Lucania, while Figure 1.20.1b the relationship between erosivity and the modified Fournier index.

The better performance of the annual precipitation amount in predicting EI_{30} confirms the direct proportionality between the two parameters and the assumptions made by Van der Knijff *at al*. (2002), even if the coefficient of proportionality is close to 2.5 (Figure 1.20.2). Using an expanded data set, which includes additional stations in Tuscany and Emilia Romagna regions, and examining the data in more detail reveal that the coefficients used to map erosivity in Italy were underestimated. The best fitting regression was the following:

$$EI_{30} = 3.08P - 944, \qquad r^2 = 0.768, \qquad n = 88 \tag{1.20.4}$$

Obviously, the linear relationship between annual precipitation and annual rainfall erosivity is not always verified. If we examine the data collected by D'Asaro and Santoro (1983) in Sicily, the results are very poor. They suggested as a simplified estimator of the mean annual erosivity the equation

$$EI_{30} = 517I_{24,2} + 49.4I_{1,2} - 1736 \tag{1.20.5}$$

where $I_{24,2}$ is the precipitation lasting 24 h with a return period of 2 years (mm/h) and $I_{1,2}$ is the 1-h rainfall with a 2-year return period (mm/h).

The data shown in Figure 1.20.2 are fairly scattered. The scatter is linked to the return period of given EI_{30} values. By applying a Gumbel distribution (i.e. extreme value type 1 distribution; Gumbel, 1958; Grigorten, 1963) to the Loiano (Emilia Romagna) and the Cavallina (Tuscany) stations, we obtain the results shown in Figure 1.20.3.

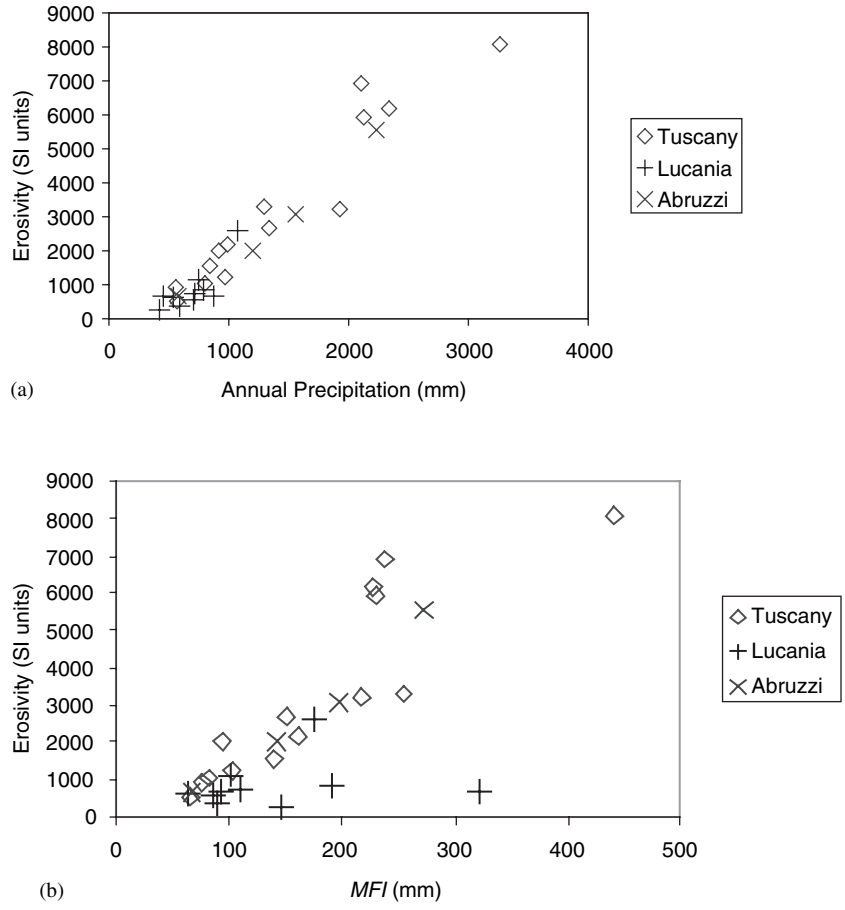

Figure 1.20.1 (a) A significant relationship exists between annual rain amount and erosivity; (b) the modified Fournier index (*MFI*) does not perform satisfactorily

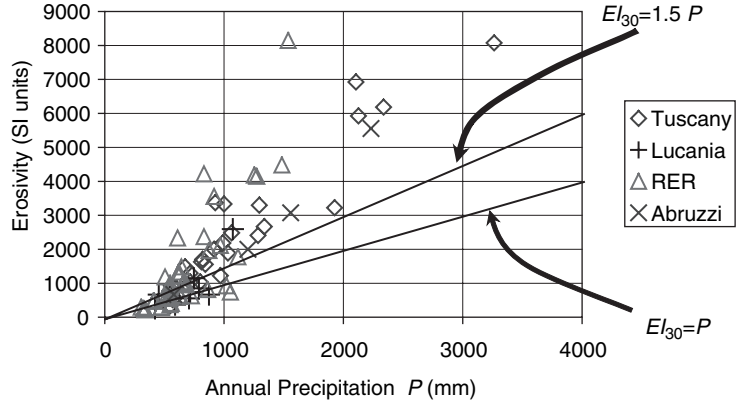

Figure 1.20.2 EI_{30} versus annual precipitation in Tuscany, Lucania, Emilia-Romagna (RER) and Abruzzi

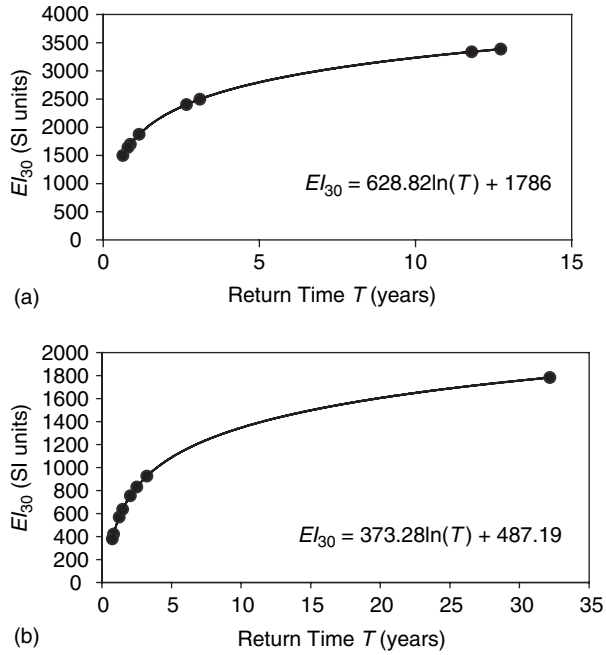

Figure 1.20.3 Annual EI_{30} versus return time, calculated with Gumbel GEV-1 distribution, for (a) Cavallina (Tuscany) and (b) Loiano (Emilia Romagna)

The data corresponding to the Loiano and Cavallina stations shown in Figure 1.20.2 are among those scattered upwards with respect to the main trend. The return periods of the calculated annual averages are actually fairly long (Table 1.20.1), which makes those values fairly rare.

Finally, investigations were conducted on the effect of climatic changes on erosivity. For a station in Valdera (Vicarello, Volterra, Tuscany), Bazzoffi and Pellegrini (1992) found the values reported in Table 1.20.2. The data indicate that the mean (over 9 years) EI_{30} value has changed since the 1960s, with a major variation of I_{30} values, which indicates precipitations with more stormy characteristics. These findings are substantially confirmed by Crisci *et al*. (2002).

1.20.2.3 Erodibility

The number of available plot studies for measuring soil erodibility is limited in Italy. The few available are reported in Table 1.20.3. Figures for the Valdera soil (Chisci G, 2003, personal communication) are only an

TABLE 1.20.1 Return period of some annual erosivity values

Location	Average EI_{30} (SI units)	Return time for average EI_{30} (years)	Extreme EI_{30} (SI units)	Return time for extreme EI_{30} (years)
Cavallina (Tuscany)	951	2.25	—	—
	—	—	3338	11.8
	—	—	3386	12.7
Loiano (Emilia Romagna)	801	2.23	—	—
	—	—	1783	32.2

TABLE 1.20.2 Variations of rain erosivity at Vicarello (Tuscany)

Period	Precipitation (mm)	Kinetic energy (MJ ha^{-1})	Erosivity (SI units)
1964–1972	604	77	852
1973–1981	711	89.6	1037
1982–1990	713	91.23	1313

Source: Bazzoffi and Pellegrini (1992).

estimate of erodibility, because they were calculated using data collected when the experimental plots were in seed-bed condition for winter wheat (mean value over four autumns). The other figures refer to 5 years of observations for Mugello (average over two plots of 100 m^2) and to 4 years for Corleto Perticara, Lucania (one plot of 60 m^2) with only 11 rainstorms that resulted in soil loss.

The three soils show remarkable temporal dynamics in their erodibility (Figure 1.20.4).

1.20.2.4 Soil Erosion and Wildfires

Studies on the effects of wildfires on soil erosion were conducted mainly by Giovannini's group (e.g. Giovannini, 1995; Giovannini *et al*., 1988). They used a series of experimental plots in S. Luce, Pisa, where they studied the effect of fires of two intensities, namely light fire with a temperature at the soil surface of about 180 °C and intense fire of about 475 °C at the soil surface. Erosion on a baseline plot under '*macchia mediterranea*' is compared with two plots where the *macchia mediterranea* was burned at the two fire intensities, and a plot where the *macchia* was removed and the soil was kept bare by cutting the sprouts (not to be mistaken for 'continuous fallow'). Results are shown in Table 1.20.4, and are subdivided into trimesters to provide a measure of the time dimension.

Analysis of the soil characteristics reveals a decrease in topsoil porosity, clay and organic matter content, with a corresponding enrichment in sand. This effect was more pronounced in the plot burned by intense fire. The change indicates a considerable modification of soil characteristics that affect soil erodibility. The tendency towards increased erodibility is confirmed by the amount of eroded soil recorded in the burned plots.

TABLE 1.20.3 Erodibility values measured in Wischmeier-like plots

Locality	Source	Soil (USDA classification)	Clay (%)	Silt (%)	Sand (%)	Organic matter (%)	Rock fragment (%)	Erodibility[a] (SI units)
Mugello, Tuscany	Zanchi (1988)	Fluvaquents	22	40	38	1.8	<5	0.005
Mugello, Tuscany	Zanchi (1988)	Eutrochrepts	54	41	5	1.6	0	0.027
Valdera, Tuscany	Chisci G (2003, personal communication)	Vertic Xerorthent	15	43	42	0.5	0	0.001
Corleto P., Lucania	De Franchi *et al*. (1983)	Ustorthent	45.5	20.6	33.5	0.7–1.5[b]	14	0.0012

[a]From mean monthly value.
[b]From data for nearby plots under arable use.

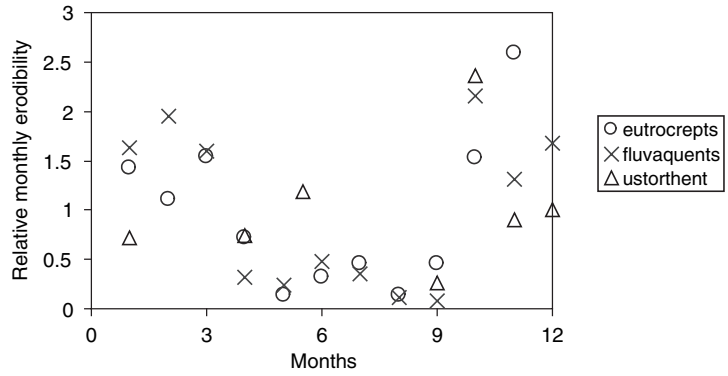

Figure 1.20.4 Seasonal variation of soil erodibility shows a cosine-like behaviour

1.20.2.5 Soil Loss During Extreme Events

Soil erosion peaks under specific land uses, such as arable land. Torri *et al*. (1999) reported 126.2 Mg ha^{-1} of soil loss during one rainstorm on two experimental plots, in seed bed condition, on the same soil as reported in Table 1.20.3 as the least erodible (vertic xerorthent). The rainstorm causing the loss was characterized by a total erosivity of 802 MJ mm ha^{-1} h^{-1}, comparable to the mean annual erosivity of the site (see Table 1.20.2). Vineyards, with rows frequently oriented downslope and with bare inter-rows, are among the main sources of sediments. Tropeano (1983) in 1981 measured more than 7 mm (i.e. about 100 Mg ha^{-1}) of soil loss in one year of erosion in vineyards sloping between 30 and 40% in Monferrato and the Langhe of Piedmont. It should be noted that the year 1981 was characterized by the lack of extreme rainfall events in the Piedmont region.

Similar values have been observed as a consequence of heavy rainstorms that occurred on 11 May 2002 near Zocca (Bologna) in the Emilia Romagna region. In this area, a severe rainstorm with a peak of 85.9 mm h^{-1} in 60 min and a total rainfall of 119.5 mm in 6 h resulted in a very intense muddy flow with removal of large part of the Ap soil horizon in the fields scarcely protected by vegetation. In addition, removal of the asphalt by intense gullying, piping along the roads and intense sedimentation in the areas of low slope gradient (e.g. gardens, roads and garages which acted as sediment ponds) were observed. In November 1987, a similar event, that took place over an area of about 30 km^2, was observed near Montalto di Castro, in northern Lazio, where the Ap soil horizon was removed in many places and the whole area was severely gullied.

The cost of these events was certainly high. As an example, the Regione Toscana, with a regional resolution (Regione Toscana, 2002a) provided €5 750 000 as a contribution for the damage produced by a downpour that hit the districts of Massa Carrara and Lucca on 20 and 21 October 1999 and the snowfall (and snowmelt) with floods that occurred in the districts of Firenze, Prato, Lucca, Pistoia, Pisa and Livorno on 18 and 19 November 1999. In another occasion, the Regione Toscana (2002b) contributed €32 138 000 to the various Tuscan districts.

TABLE 1.20.4 Soil loss (kg ha^{-1}) measured since date of experimental fires

Trimester	Rainfall (mm)	Macchia mediterranea	Bare soil	Light fire	Intense fire
I	87	1.0	3.0	20.0	443.0
II	180	1.5	1.5	2.5	86
III	190	1.5	1.5	63.5	447.0
IV	562	27.0	28.0	58.0	496.0
Year's total	1019	31.0	34.0	144.0	1472.0

Data from Giovannini (1995).

TABLE 1.20.5 List of some of the most significant events with abundant landslides, significant floods and associated soil erosion in the last 25 years in Italy

Geographic area reference[a]	Date	Trigger	Extent
Valtellina, Lombardy Region (1)	17–19 July 1987	Prolonged, high-intensity rainfall	Regional
Calabria Region (2)	December 1990	Prolonged rainfall	Regional
Piedmont Region (3)	2–6 November 1994	Prolonged, high-intensity rainfall	Supra-regional
Versilia, Tuscany Region (4)	18 June 1996	High-intensity rainfall	Localized
Crotone, Calabria Region (5)	14 October 1996	High-intensity rainfall	Localized
Central Umbria Region (6)	January 1997	Rapid snowmelt	Regional
P.zo d'Alvano, Campania Region	5 May 1998	Rainfall	Localized
Valle d'Aosta and Piedmont Regions	13–15 October 2000	Prolonged, high-intensity rainfall	Supra-regional
Imperia district, Liguria Region (7)	23 November 2000	Prolonged, high-intensity rainfall	Localized

[a](1) Crosta *et al*. (1990); (2) Antronico *et al*. (1991); (3) Regione Piemonte (1998); (4) Rosso and Serva (1998); (5) Gabriele (1997); (6) Cardinali *et al*. (2000); (7) Guzzetti *et al*. (2004).

Table 1.20.5 summarizes the most important events for which data exist in the last 25 years. Areas in Italy more subject to water erosion are outlined in Figure 1.20.5.

Values of soil loss measured over many years are difficult to find. Some, relative to sedimentation in reservoirs, were published by Bazzoffi (1987) and Van Rompaey *et al*. (2003). When plotted against the

Figure 1.20.5 Areas where water erosion features can be easily observed (readjusted, after De Ploey, 1989)

catchment size, they reveal that the main source of sediment is the channel system rather than the slope. The few catchments where major soil use was arable and the effect of erosion along permanent channels was zero or scarce indicate deposition rates ranging between 8.4 and 3.4 Mg ha^{-1} yr^{-1} (watershed area ranging between 82 and 350 ha). It is evident that soil loss in Italy is due to a limited number of events. The situation of Vicarello, in Valdera (Tuscany), illustrates this clearly. Low erodibility, medium value of the mean annual erosivity and low sedimentation values in a nearby reservoir (8.4 Mg ha^{-1} yr^{-1}; Bazzoffi and Panicucci, 1983) characterize the site. Nevertheless, a single rainstorm was enough to produce 126 Mg ha^{-1} of soil loss. The same conclusion was drawn by Roggero and Toderi (2002), with results from experimental plots in Agugliano, in the Marche region, where 60–80 Mg ha^{-1} of soil loss were observed in a few summer rainstorms under sunflowers.

1.20.3 FLOODS AND LANDSLIDES AS GEO-HYDROLOGICAL RISKS

Among the effects of climate and land-use changes are the changes in the frequency and magnitude of geo-hydrological events, such as floods and landslides. Natural causes of floods include intense or prolonged rainfall and rapid melting of snow. Floods can also be human induced. The failure or overtopping of artificial or natural dams, the collapse of artificial lakes and the overtopping or failure of artificial or natural levées can result in sudden and catastrophic inundations. In the Mediterranean area, flash floods form very rapidly, usually over a period of 0.5–2 h, as the result of localized, high-intensity rainstorms. Floods along major rivers (e.g. the Po) last from a few to several days, and are caused by prolonged and intense rainfall events over large regions, most commonly associated with characteristic and recurrent meteorological conditions. An overview of the data on sediment transport along rivers in Italy is given in Table 1.20.6.

Landslides are triggered by a variety of natural and human-induced causes. Natural causes of landslides include intense or prolonged rainfall, rapid melting of snow and earthquakes. Human-induced causes of slope failures include deforestation, unsuitable agricultural practices, forest fires, construction of artificial lakes and channels, and overloading of the top or undercutting of the toe of natural or artificial slopes. Both dimensions and velocities of mass movements span many orders of magnitude. Mass movements can occur singularly or in groups of up to several thousands. Multiple landslides occur almost simultaneously when slopes are shaken by an earthquake or over a period of hours or days when failures are triggered by intense rainfall or snowmelt. Landslides can involve flowing, sliding, toppling or falling movements, and many landslides exhibit a combination of these types of movements. The extraordinary breadth of the spectrum of landslide phenomena makes it difficult to define a single methodology to study landslides and to evaluate the associated hazards and risk.

Floods and landslides cause casualties and economic losses which are larger than, or comparable to, those produced by other damaging natural events, including earthquakes, hurricanes and volcanic eruptions. In Italy in the period 1900–2002, landslides have caused at least 5278 fatalities, comprising 5190 deaths and 88 missing persons. In the same period, floods have caused at least 2750 fatalities, comprising 2630 deaths and 120 missing persons (Guzzetti, 2000). These figures correspond to 8.5 landslide events with fatalities and 7.3 flood events with fatalities per year, in the investigated period.

In Italy, investigations aimed at collecting bibliographic and archive information on landslide and flood events in the period 1900–2002 allowed the compilation of a large database of historical and recent geo-hydrological events with consequences. In spite of the limitations due to the complexity of the Italian territory, of the different awareness of the impact of landslides and floods on the territory and of the limited resources available for the inventory and its maintenance and upgrade, the result of the inventory represents the most comprehensive source of information on mass movements and floods ever prepared for Italy. The database contains 31 265 records of 31 000 landslide events at 21 400 sites (equivalent to a density of one landslide per 15 km^2) and 8520 records of 30 468 inundation events at 13 494 sites (equivalent to a density of one inundated site per 25 km^2). Stored in the database is information obtained from about 100 000 newspaper articles and 2000 scientific and technical reports. Most of the information is readily available on the Internet (http://sici.irpi.cnr.it).

TABLE 1.20.6 Solid discharge values in Italy: minimum, maximum and monthly and yearly average values for 41 gauging stations with at least 15 years of measurements

River and station	Area (km^2)	Years	Solid discharge (Mg/km^{-2})			
			Min.	Max.	Monthly av.	Yearly av.
Po at Pontelagoscuro	54290	30	0.0	136.9	15.8	186.7
Po at Boretto	44070	27	0.1	113.5	17.0	198.7
Po at Piacenza	35430	28	0.0	116.3	7.6	90.3
Tevere at Roma Ripetta	16545	34	0.2	232.5	21.4	238.9
Adige at Boara Pisani	11954	25	0.1	52.8	7.0	83.6
Arno at San Giovanni alla Vena	8186	40	0.0	251.6	27.2	327.0
Po at Meirano	4885	19	0.0	247.7	6.0	71.6
Adige at Trento	4597	26	0.1	321.1	19.0	228.4
Arno at Nave di Rosano	4083	34	0.0	622.1	20.0	236.8
Pescara at Santa Teresa	3125	20	0.2	598.4	35.2	422.5
Bradano at Tavole Palatine	2743	34	0.0	3999.4	66.2	794.3
Ofanto at San Samuele	2716	31	0.0	813.7	18.8	220.8
Ombrone at Sasso d'Ombrone	2657	38	0.0	948.4	38.7	463.9
Volturno at Amorosi	2015	16	0.1	665.0	53.9	646.3
Imera meridionale at Drasi	1782	24	0.0	2973.9	61.5	738.1
Crati at Conca	1332	31	0.0	1439.4	60.7	720.9
Canale Maestro della Chiana	1272	39	0.0	91.2	5.1	60.7
Sinni at Valsinni	1142	26	0.0	1294.6	79.1	949.7
Reno at Casalecchio	1051	29	0.0	1132.3	87.2	1046.9
Ofanto at Moneteverde	1028	26	0.0	1644.0	32.3	382.9
Magra at Calamazza	939	15	0.0	475.0	42.9	514.8
Arno at Subbiano	738	38	0.0	901.1	24.9	298.3
Savio at San Vittore	597	27	0.0	1318.3	123.3	1479.0
Orcia at Monte Amiata	580	28	0.0	1562.1	86.3	1035.3
Era at Capannoli	337	18	0.0	65.5	4.9	58.3
Ofanto at Cairano	272	20	0.0	326.1	14.4	173.1
Senio at Castel Bolognese	269	27	0.0	814.1	67.7	812.8
Venosa	261	15	0.0	685.8	17.6	210.7
Atella at Ponte Sotto Atella	175	34	0.0	727.8	29.3	351.5
Arcidiaconata	124	32	0.0	1782.3	34.1	406.9
Vulgano at Ponte Troia-Lucera	94	17	0.0	983.0	26.2	314.2
Celone at San Vincenzo	86	16	0.0	275.1	18.5	221.6
Anapo at San Nicola	82	15	0.0	286.2	5.4	64.8
Canale Santa Maria	60	15	0.0	438.1	13.5	147.5
Lese at Schiena d'Asino	60	18	0.0	5.5	0.1	1.0
Triolo	54	18	0.0	626.4	22.0	246.6
Eleuterio at Risalaimi	53	19	0.0	192.8	7.9	94.5
Casanova	52	16	0.0	428.3	15.6	186.7
Salsola at Casanova	43	16	0.0	658.1	19.7	236.9
Trionto at Difesa	32	15	0.0	11.7	0.2	1.8
Alaco at Mammone	15	18	0.0	15.5	0.6	6.9

For the Umbria region, an area that extends for 8456 km^2 in central Italy, detailed landslide inventory maps are available (Guzzetti *et al*., 2002). Such landslide maps were compiled through the interpretation of single and multiple sets of aerial photographs, at nominal scales ranging from 1:13 000 to 1:35 000. The reconnaissance inventory reveals that landslides cover about 8% of the entire territory. Landslide density increases to 10% if large valley bottoms and intra-mountain basins are excluded from the analysis. In places, landslide density can be even higher, exceeding 20%. Comparing in a GIS the landslide inventory map and the available land-use map, at 1:10,000 scale, showed the land-use types most affected by landslide phenomena, at the regional scale. The analysis revealed that landslides in the Umbria region are most abundant in forested (8.81%) and in cultivated areas (8.31%). A difference exists between the prevalent landslide types in the two broad land-use categories. In forested areas, landslides are mostly old and very old, and relict or dormant. Only a few recent and active landslides were identified in forested areas. In cultivated areas, old and very old landslides coexist with recent and active slope failures of various types.

An opportunity to investigate the relationships between active landslides and land-use types and practices in Umbria arose in January 1997. In December 1996, a large snowstorm covered the Umbria region of central Italy with a thick snow cover. A sudden change in temperature occurred on 31 December and melted most of the snow, triggering thousands of shallow and deep-seated landslides. Most of the slope failures occurred in the period 3–7 January 1997. Field surveys, helicopter flights and 3-month-later aerial photographs allowed landslide mapping over about 1500 km^2. The entire inventory map covers about 2000 km^2 and contains 4233 landslides. Snowmelt-induced slope failures were mostly shallow soil-slips (53%) and slump earth-flows (9%). Deep-seated failures (38%) comprised complex or compound movements. The total landslide area is 12.7 km^2 (0.6% of the study area) with an average density of 2.1 landslides per km^2.

An attempt was made to investigate the existing relationships between slope failures and local agricultural practices based on this snowmelt-induced landslide inventory. Owing to complex and stringent economical and social issues of European agriculture, the topic is of major relevance. During the past 20 years, EU legislation has encouraged and subsidized major changes in land use throughout Europe. In the hilly and low mountainous areas of Italy, EU agricultural policies led to a progressive increase in arable land for cereal (mainly wheat) production in areas traditionally mantled by shrub and low forest, intended for pasture or abandoned. Furthermore, because of the advancements in earthwork machinery, the new agricultural practices were extended on to land surfaces with slope angles frequently greater than 25° and locally greater than 45°. Most of these areas had never been ploughed before.

The impact on the stability of the slopes and river channels of such a major land-use change appears poorly investigated by the interested parties (e.g. Ministry of Agriculture, Ministry of Environment). At present, no comprehensive and quantitative data exist on the influence of such a transformation on the stability conditions at the basin scale in Italy. In spite of the lack of investigations, scattered local sources of information and field investigations in the Umbria region completed after the January 1997 event and in subsequent studies indicate a generalized increase in shallow landslides in areas that had undergone land-use changes in recent years, chiefly clear cutting and deep ploughing. Simple buffering and map overlay operations in a GIS revealed that about 75% of all new failures occurred inside or within 150 m of an already existing landslide, confirming the spatial persistence of landslide phenomena in the Umbria region (Cardinale *et al*. 2000).

1.20.4 MECHANICAL SOIL EROSION

1.20.4.1 Tillage Erosion

Soil translocation due to tillage is particularly intense in Italy because of the Italian dominant morphology and because the ploughing depth is often greater than 40 cm. The long-term effect of tillage is that of smoothing the

landscape and of sculpting it at field borders, producing steps and discontinuities which often resemble grassed terraces. This soil removal on hillslope convexities leads to significant and possibly adverse changes in soil properties. This is often visible in the fields as patches of soils of different colours (often darker where the soil is still preserved or where accumulation prevails). Such changes will affect soil quality and productivity and may also influence water and wind erosion rates by exposing a more erodible subsoil (as is often the case in Italy).

Values from extensive surveys in Tuscany (Borselli *et al*., 2002) indicate that tillage erosion is often close to 2 cm yr^{-1} of surface lowering with peaks in convex spots up to 4 cm yr^{-1}. A better situation is instead that in the pre-Apennines near Bologna, where tillage translocation rates are usually smaller, often close to 1 cm yr^{-1} of surface lowering.

1.20.4.2 Land Levelling

Land levelling is applied to undulating land to adapt the ground surface to mechanized agriculture. Bulldozers are used to remove the natural vegetation, the remains of old cultivations and all the unwanted undulations, including old terraces. Land levelling is usually used for tree-crops (i.e. mainly vineyards, olive groves and orchards), but arable land is also often levelled. Figure 1.20.6 shows the extent of the areas most affected by land levelling in Italy.

Figure 1.20.7 shows an example of land levelling and of its interaction with rainfall (i.e. gullies). Land levelling is usually carried out in summer so that the levelled slope will be prone to high erosion rates in autumn when rainfall peaks. At Cesena, in the Emilia Romagna region, Bazzoffi *et al*. (1989) measured 454 Mg ha^{-1} of water erosion. This figure is confirmed by independent observation in the Chianti region of

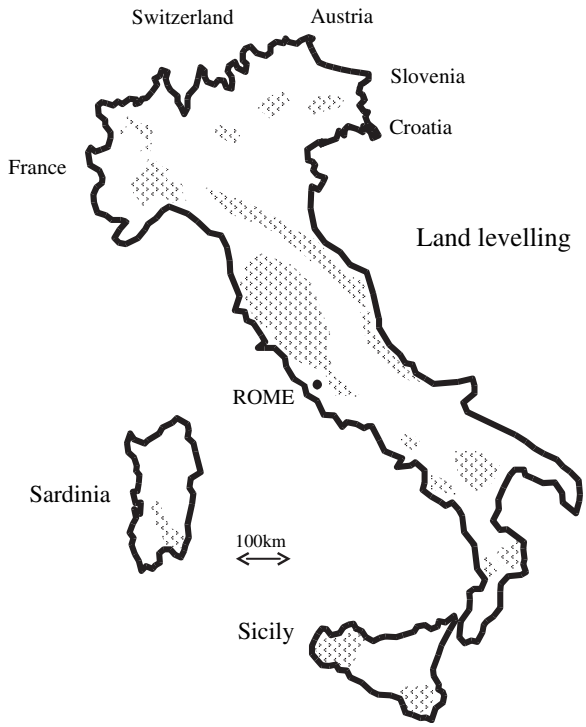

Figure 1.20.6 Areas where land levelling is more likely to be used

Figure 1.20.7 Land levelling in Tuscany: note the stone walls made using boulders removed from the field. The whole field is severely gullied and rilled (spring 2003)

Tuscany (Borselli L, Torri D, unpublished data) in terms of gully and rill excavation that occurred in a period of 1 year. In addition, landslides associated with land levelling are common, owing general destabilization of the levelled field and of the contiguous upslope field, confirming the hypothesis proposed by Torri *et al*. (2002).

No national legislation exists on land levelling in Italy. Every region has passed its own legislation. For example, in Tuscany permission must be requested when the scalping affects a depth greater than 50 cm, in which case a hydro-geological report is required, certifying that slope stability and slope hydro-geology are unaffected after the levelling. From a technical point of view, no advice is given for land levelling, which is made following various procedures. Often the only aim is that of decreasing the slope gradient, obtaining a smoother slope surface. This can be done (1) on the field as it is, (2) after having 'sieved' and remixed the upper 1–1.5 m of soil, (3) after the removal of large stones and boulders, (4) with the removal of the upper 0.5–1 m of soil in order to redistribute it on the levelled surface and (5) by various combinations of the previous methods. Locally the soil may be removed entirely.

1.20.5 BADLANDS

In Italy, several geomorphological and lithological settings are characterized by badlands, locally known as '*calanchi*' or '*biancane*' (see Torri *et al*., 2000; Rodolfi and Torri, 1997; and references therein). '*Calanchi*' are described as small hydrographical units, with steep and eroded slopes, locally subdivided into several secondary valleys, separated by more or less sharp ridges. '*Biancane*' are dome-shaped forms, generally less than 20 m in height, locally isolated but more often grouped in fields ('*campi*'). In Tuscany, '*biancane*' are characterized by bare, south-facing slopes, with the north-facing slope covered by scanty herbaceous vegetation. In southern Italy, the effect of the topographic aspect on the '*Biancane*' is less distinct or completely lacking, and the domes are mostly bare of vegetation.

Values of retreat rates for the '*calanchi*' headcuts have never been properly measured, and the few measurements available are often lost in local papers. Field evidence and the interpretation of aerial photographs reveal that erosion rates vary considerably. Rates are locally close to a few decimetres per year, but values of about 1 m yr^{-1} were observed in the badlands near Sestino, in the Arezzo Province of south-east-Tuscany (Ch. Iasio, personal communication). In Lucania (southern Italy), Alexander (1982) measured

Figure 1.20.8 Locations where badlands are frequent and active (based on maps by Bucciante, 1922, and De Ploey, 1989)

denudation rates close to 2 cm yr^{-1}. In Tuscany (central Italy), Colica (1992) measured erosion rates of 3–4 cm yr^{-1} in the south-facing slopes and of 2 cm yr^{-1} in the south-west- and south-east-facing slopes.

Since the map drawn by Bucciante (1922), the extent of badlands has been reduced, owing to reclamation work undertaken especially in the past three decades (Moretti and Rodolfi, 2000, in Abruzzi; Rodolfi and Torri, 1997, in Tuscany; Clarke and Rendell, 2000, in Lucania). Locations where badlands are more common in Italy are shown in Figure 1.20.8.

1.20.6 SOIL CONSERVATION MEASURES

During the 19th century a wide variety of hill-conservation measures to prevent runoff and erosion were designed, especially in Tuscany, and then spread over the entire Italian territory. Among of the many agro-hydraulic conservation systems, the more representative are the following:

1. *Contour ditching*, constituted by ditches, which closely follow contour lines, and drain into protected waterways. The distances between the ditches are about 4–5 m.
2. *Step Terracing*, adopted on steeper slopes. It consists in setting up small terraces using, for the step building, the stones removed from the field.

3. *Grassed-wall and stone-wall bench terraces*, i.e. small terraces in which ditches or underground drainage collects surplus water in protected waterways.
4. *'Pensile' soil platforms*, adopted along rocky, steep slopes where olive trees are planted on a soil platform rounded below by a semicircular stony wall, built with the aim of keeping and protecting a sufficient quantity of soil.

In many places, it is still possible to see impressive agro-hydraulic structures (e.g. the spectacular terraces that characterize the landscape of the Cinque Terre, in the Liguria region). Their maintenance was based on a different agricultural economy. Nowadays, these systems are kept up only in areas of particular interest.

Starting in the 1950s, the industrialization of agriculture and the need for cost reductions and field enlargement for mechanical cultivation led to a widespread removal of these conservation structures. The remaining ones are usually not maintained, so they are often in a condition of severe and continuous degradation. The evolution towards new systems of soil and water conservation, more suitable to modern agriculture, maintains the basic concept of removal of surplus waters through contour ditches and tile drains.

On more gentle slopes, *contour ditches*, 60–100 m apart, represent a new system for providing sufficiently large surfaces for machinery. The ditch's depth is set lower than the plough sole, so that sub-surface drainage is guaranteed. Ditches are often substituted by counter-slope field roads, which facilitate the movement of mechanical equipment, and water drainage, In the new system, water is drained by ditches into protected waterways. The enlargement of the slope length has been made possible by an increase in the ploughing depth. In fact, the more consistent volume of ploughed soil results in an increased water-storage capacity, with a corresponding reduction in superficial runoff, especially during the more aggressive rainstorms in the autumn when soil is cloddy.

In the sandy soils of the Asti province of Piedmont region, the *linked bench terracing* system has been proposed for valuable vineyards on sloping surfaces. In this system, terraces follow a jigsaw pattern up and down the slope, allowing continuity between subsequent planes.

In recent years, some conservation measures have been tested that involve grass-mulch cover management in conjunction with lower and more precise inputs of nutrients. These grass protection systems are going to be adopted in vineyards and orchards, in the inter-row space. Their contribution to soil loss reduction and fertility protection, without reducing the quantity and quality of the products, depends on the interaction between agronomic management and climate, which are not always easy to control.

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1.21

Albania

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1.21.1 GENERAL CHARACTERISTICS

Albania is located in the south-western part of the Balkan Peninsula, between 39° 38′ and 42° 39′ N, 19° 16′ and 21° 40′ E. Only 14% of the country has a gradient of less than 8%, 53% has a gradient of 8–30% and 33% is steeper than 30%. It is very mountainous, with a great variety of landscapes including bare rocks. The average elevation above sea level is 786 m with an average slope of 25%.

Agriculture is practiced throughout the country, but is most intensive in the western coastal lowland. About 40% of agricultural land has a gradient <8%, 49 % has a gradient of 8–25% and 11% is steeper than 25%.

The geographical position of Albania results in a Mediterranean climate for the western lowland coastal area and a continental climate in the remaining areas. The average annual temperature is 12.9 °C with a maximum in July of 21.8 °C and a minimum in January of 4.1 °C. Although Albania is a small country, rainfall is highly variable in time and space. The rainfall regime is of the Mediterranean type with a dry summer season and a rainy season in autumn and winter. The average rainfall is 1400 mm yr^{-1}. The precipitation falls especially in autumn (558 mm or 40% of the total) and winter (362 mm or 26%), with a maximum in October (271.5 mm or 17.5%) and minimum in June (35 mm or 2.5%). These high rainfalls present a severe risk in generating runoff and sediment loss. Rondelli *et al*. (1995) have calculated a high value of the erosivity index for Albania (annual mean value 369.7).

The total land area of Albania is 2.875×10^6 ha, of which 1 026 000 ha is forest, 445 000 ha is pasture, 700 000 ha is cultivated (580 000 ha arable and 120 000 ha tree crops and vineyards) and 704 000 ha has other uses (urban areas, roads, etc.) (Shallari *et al.*, 2002). Pine, oak, beech, juniper and cedar are dominant species in natural forests. Olive, citrus, vineyards, apple and plum are the principal planted fruit tree species. Sheep fescue is widespread, in addition to Gramineae. The dominant cereals are wheat, barley and maize.

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Most of the arable land occurs on mainly flat, or gently sloping, flood plain and river terrace areas. Soils are mostly derived from river alluvium and colluvial wash deposits and are deep, calcareous, medium textured and potentially very fertile, with generally high water retention and nutrient retention capacities. Smaller areas in more hilly areas show more of a mixture of colluvial and alluvial materials and stony soils are more common. Uneven topography and variability in soils, notably surface texture and infiltration rate, are the dominant problems.

Swamp areas, reclaimed mostly in the 1950s and 1960s, have large extents of fine-grained and peat soils. Most of these areas are virtually at sea level and depend on pumped drainage for their successful exploitation. After drainage and reclamation, however, much of the peat has oxidized, with the result that land surfaces are lower by up to 2–3 m compared with their original elevations.

The map of soil erodibility class shows that most of Albania belongs to moderate and strong erodibility classes (Grazhdani, 2003). This means that the soils are highly erodible. This is most evident in the southern and northern parts of Albania.

Albania has a sparse land cover (Grazhdani, 2003). In autumn and winter, rainfall peaks at an average of 920 mm (66% of the total). Minimum land cover is in October and November. In this period, about 32% of the country has a land cover $<10\%$. There is a land cover of 90–95% for 90% of the country for only 2 months (June and July). Vegetation does not play an essential role in soil erosion in this season, because during June and July rainfall reaches only 85 mm (6% of the total).

Soil water storage is very low in October and November, which means that strong erosion is expected in these months. Soil water storage in October is <40 mm for 80% of the country, 41–80 mm for 12% and >80 mm for 8%. In November, soil water storage is <40 mm for 76% of the country, 41–80 mm for 14% and >80 mm for 10%. A contrasting situation is seen in June and July with storage, which means that low erosion is expected. Storage is <40 mm for 13% of the country, 41–80 mm for 60% and >81 mm for 27%.

1.21.2 HISTORICAL EVIDENCE FOR EROSION

Soil erosion in Albania accelerated when the people chopped down trees and began to farm the sloping lands of the northern and eastern parts of the country. Soil erosion was a factor in the declining productivity of these lands, which, in time, led to their abandonment and the westward migration of people in search of new farmlands. It was not until the early 1930s that King Zog recognized the damage being done and obtained government support for erosion control efforts.

After World War II, high population growth led to expansion of the area of land under cultivation, clearing and burning steep, forested slopes and ploughing grasslands. Intensified cultivation of fertile, relatively level lands has helped produce much of the needed food. Population pressure also led to overgrazing of rangelands and over-exploitation of timber resources. All these activities degraded or removed natural vegetation, causing the underlying soils to become much more susceptible to erosion.

The declining productivity of farm, forest, and rangelands tells only part of the sad erosion history in Albania. The environmental and economic damage suffered by sites on which the eroded soil materials are deposited is as great as or greater than that incurred on the sites from which the soil material was removed.

A little progress in reducing soil erosion was made during the communist regime when such physical practices as contour strips, terraces and windbreaks were installed. Then, in the decade from 1992 to 2002, substantial progress was made in reducing soil erosion, largely as a result of two factors: the spread of conservation tillage and the implementation of land-use changes. A major part of the reduction in soil erosion experienced in Albania since 1992 is due to government programmes that have paid farmers to shift some land from crops to grass and forests (150 000 ha). Establishing grass or trees on the former cropland reduced sheet and rill erosion from an average of $6\,\mathrm{t\,ha^{-1}\,yr^{-1}}$ (Zdruli, 1995). Progress continues to be made on both fronts.

However, about one-third of the nation's cultivated cropland is still losing more than $10\,t\,ha^{-1}\,yr^{-1}$, the maximum loss that can be sustained without serious loss of productivity on most soils.

Now, after some 80 years of soil conservation efforts, soil erosion is still a prominent problem on about half of the total cropland area of the country. The current threat of soil erosion is more ominous than any time in history.

1.21.3 CURRENT SOIL EROSION

The current status of soil erosion processes is severe in Albania owing to a number of specific physical and socio-economic conditions. The main physical factors influencing these processes are its irregular terrain and steep slopes, frequent periods of drought and heavy rainfall and the presence of high erosion-prone soils owing to their weak structure, shallowness and lack of organic matter. The main socio-economic factors influencing soil erosion are destruction of forest to make soils available for pastures and crops, overgrazing of rangeland, bush and forest fires, causing extensive loss of natural vegetation, and over-exploitation of agricultural land inducing soil erosion.

In recent decades, new trends in farming and agricultural economics led to deep changes in farming practice, stimulating soil erosion processes. Some examples of these changes are the following: increasing mechanization, which induces soil compaction and water runoff; lack of maintenance or degradation of terraces and ditches; destruction of field boundaries (hedges, ditches, stone walls, tree rows) and excessive land levelling; monoculture and continuous cropping; reduced rate of manure application; excessive tillage and removal of crop residues; and lack of adequate contour ploughing and cropping.

In order to estimate accumulative soil erosion we (Grazhdani, 2003) have used the Regional Degradation Index (RDI) with the Climatic Soil Erosion Potential (CSEP) concept (Kirkby and Cox, 1995). The model uses existing soil maps, land use maps, a digital elevation model (1 km), and interpolated climate data. The daily rainfall sequences for the period 1980–2000 are obtained for 23 climate stations distributed throughout the country. Maps of annual and monthly predicted soil erosion rates are then compiled at the School of Geography, University of Leeds, UK.

The erosion maps clearly show that Albania is a country where erosion is potentially severe. Annual erosion rates are especially high in the southern and central areas of the country. There are three areas where the annual erosion rate is more than $300\,t\,ha^{-1}\,yr^{-1}$: two areas in Gjirokastër (510 and $380\,t\,ha^{-1}\,yr^{-1}$) and one in Sarandë ($460\,t\,ha^{-1}\,yr^{-1}$). The highest annual erosion rates of more than $10\,t\,ha^{-1}\,yr^{-1}$ are in Gjirokastër, Sarandë, Vlorë, Durrës, Kavajë, Tiranë, Lezhë, Shkodër and Skrapar. Annual erosion rates are the lowest annual erosion rates of generally less than $10\,t\,ha^{-1}\,yr^{-1}$ are in Tropojë, Pukë, Kukës, Korçë, Gramsh, Fier and Lushnjë. Erosion rates are very high especially in October, November, February and December and low in June and July. These rates show that soil erosion constitutes a severe problem in Albania and that erosion in most cases is not well controlled and the appropriate soil management practices are not being applied. This evidently creates not only environmental but also socio-economic costs. In Figure 1.21.1 are shown the hotspots of water soil erosion in Albania.

Albania has a very dense hydrographic network made up of streams, rivers, lakes and the Adriatic and Ionic Sea. The main rivers of Albania have the same orientation as the range of mountains. They flow from south-east to north-west or from east to west to the sea. The main watersheds that drain to the Adriatic Sea and their sediment yields are shown in Table 1.21.1.

All watersheds exhibit excessive sediment transport that fills reservoirs or aggrades the channel, on the flood plain, in irrigation and drainage structures or in developing delta areas. The analysis indicates that the most significant factor contributing to the increased flooding and sediment problems is river instability, which in turn is caused by gravel mining. The gravel extraction itself creates a hole in the river channel that may be

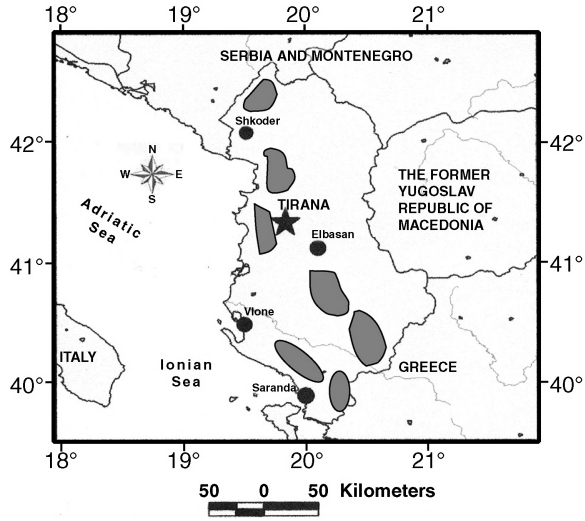

Figure 1.21.1 Hotspots of water soil erosion in Albania

3–5 m deep. This accelerates sediment movement, some of which replenishes the gravel supply being extracted. In a simulation of River Shkumbini morphology (Molinas, 2002), the rate of sediment transport is about two orders of magnitude greater than the annual mean value without extraction. Once the down cutting is accomplished, the river moves laterally as a means of adjusting to the new energy gradient. The lateral movement or accelerated meandering undermines and erodes adjacent riverbanks, causing a loss of land adjacent to the river. The problem is becoming more critical in the lower reaches of the river, approaching the Adriatic Sea. The evidence is the delta being built at the mouth of the river and a rise in the base elevation of the river. As a result, the average river gradient has decreased, causing further aggradation and exacerbating the problem. The degraded river is compensating for the new profile by meandering or moving laterally and cutting banks and causing more erosion. All of this contributes to the increase in flooding and sediment yield that is perceived to be happening and causing other river responses. For example, the mouth of the Shkumbini River has moved northwards by 4.5 km in the 14-year period from 1986 to 2000, cutting off several kilometres of river. In addition, the Shkumbini River is developing a delta at the confluence with the Adriatic Sea that was not present prior to 1986.

TABLE 1.21.1 Average sediment yield of main rivers in Albania, 1975–2000

River	Watershed area (km^2)	Average slope of the basin (%)	Average slope of the river (%)	Sediment yield ($t\,ha^{-1}\,yr^{-1}$)
Drini	5 973	26	5.0	18.8
Mati	2 441	35	0.8	9.32
Fani	1 076	34	1.1	11.1
Erzeni	760	26	6.0	26.6
Shkumbini	2 444	30	6.2	23.7
Semani	5 649	32	3.6	20.9
Vjosa	6 706	28	4.1	13.5

1.21.4 ON-SITE AND OFF-SITE PROBLEMS AND COSTS

Erosion damages the site on which it occurs and also has undesirable effects off-site in the larger environment. Although the costs associated with either or both of these types of damage may not be immediately apparent, they are real in Albania and grow with time. Landowners and society as a whole must eventually foot the bill.

The most obvious damaging aspect of erosion in Albania is the loss of soil itself. In reality, the damage done to the soil is greater than the amount of the soil lost because the soil material eroded away is almost always more valuable than that left behind. Experiments have shown the organic matter and nitrogen contents in the eroded material to be higher than those in the original topsoil. The soil left behind usually has a lower water-holding capacity and cation-exchange capacity, less biological activity and a reduced capacity to supply nutrients for plant growth. The deterioration of the soil structure often leaves a dense crust on the soil surface, which, in turn, greatly reduces water infiltration and increases water runoff.

Erosion moves sediment and nutrients off the land. Sediment deposited on the stream bottom can have a disastrous effect on many freshwater fish by burying the pebbles and rocks among which they normally spawn. The buildup of bottom sediments actually raises the level of the river, so that flooding becomes more frequent and more severe. In the Lezha and Elbasan areas, catastrophic flooding has become common in the last decade.

A number of major problems occur when sediment-laden rivers reach a lake, reservoir or estuary. Here the water slows and drops its load of sediment. It is estimated that about 1.6×10^6 t of sediment are deposited each year in the nation's reservoirs. Sedimentation in the sea represents catastrophic loss of soil fertility in addition to the spoiling of attractive coastlines and beaches.

The large swamp reclamation projects in the 1950s and 1960s (about 130 000 ha) have had a number of negative environmental impacts, including loss of natural floodplain areas, with the result that much more of the heavy silt load of the major rivers ends up in the sea, spoiling beaches and local fisheries. This also leads to the loss of natural wetland, of importance as a wintering ground of migratory water birds and as a breeding ground for a number of endangered species, the most notable being the Dalmatian pelican, loss of natural fisheries and areas for hunting and collecting of natural products, such as reeds, molluscs, crustaceans, ducks and geese, and massive oxidation of organic matter in peat and fine-grained soils, as the swamps have been drained. The rate of peat oxidation is of the order of 200–300 t CO_2 ha^{-1} yr^{-1}, which, if a rational international carbon tax were to be introduced, would make cultivation of such land uneconomic.

Although no precise data exist, national or regional average water erosion rates have been used to estimate the total costs of erosion in Albania. Included in such calculations are the on-site costs of replacing nutrients and water lost through accelerated erosion, in addition to crop yield reduction due to reduced soil depth. The average annual total nitrogen, potassium and phosphorus losses of 4.9, 28.64 and 3.56 kg ha^{-1}, respectively, are estimated for experimental sites located in southeastern Albania (Grazhdani *et al.*, 1996). Depending mainly on assumptions about the value of nutrient lost in sediment and runoff, the total annual on-site costs have been estimated at about US$12 million (Shallari *et al.*, 2002).

The off-site costs of erosion are likely to be even greater, especially because of the health effects and the reduced recreational (fishing, swimming and aesthetic) value of muddy waters. The total of these annual off-site costs has been estimated at about US$9 million (Shallari *et al.*, 2002). Such high costs are a sobering reminder of the burden that society bears as a result of poor land management and would seem to justify increasing the sums allocated to the battle against erosion.

1.21.5 SOIL CONSERVATION AND POLICIES TO COMBAT EROSION AND OFF-SITE PROBLEMS

Nearly 50×10^6t of soil are eroded each year on land in Albania (Troendle, 2002). Half of this erosion occurs on the nation's croplands and the remainder on harvested timber areas, rangelands and construction sites.

About one-third of the cropland still suffers from erosion that exceeds levels thought to be tolerable. Protecting soil from the ravages of water is by far the most effective way to constrain erosion.

At national level, much needs to be done on wildlife preservation and reforestation, linked to watershed management programmes. Land-use planning issues, particularly those of unplanned and unrestricted building on excellent agricultural areas, also needs urgent attention.

Agronomic effects and costs of erosion on crop yield have not been adequately assessed in Albania. Fortunately, recent decades have seen much progress in understanding the mechanisms of erosion and in developing techniques that can effectively and economically control soil loss in most situations.

According to our research, soil erosion on croplands can most readily be controlled by managing vegetation, plant residue and soil tillage. A study was conducted from 1992 to 1998 in south-eastern Albania (Grazhdani and Dhima, 1999). Mouldboard (MB) ploughing was compared with two conservation tillage methods: disking (DK) and no-till (NT). The two methods were evaluated on two crop rotations: pasture–wheat (PW) and wheat–barley–grain legume (WBL). Each plot was 25×5 m with a uniform slope of approximately 10%. The plot was bordered with sheet steel strips ($100 \times 30 \times 0.2$ cm) driven into the ground to a depth of 20 cm and an endplate at the bottom end to collect runoff. Measurements performed included some soil properties and surface cover, runoff, amount of sediment in the runoff and the concentration of nitrogen and phosphorus in sediment (Table 1.21.2). Crop–pasture systems, particularly those including forage legumes and fodder grasses, increase crop and livestock production while improving soil quality through enhanced biological activity and physical structure. Likewise, the use of conservation tillage systems, which leave most of the plant residues on the surface, greatly decreases erosion hazard. Crop rotation, coupled with such practices as contour tillage, strip cropping and terracing, also helps to combat erosion on farmland.

Undisturbed forests and dense grass provide the best soil protection and are about equal in their effectiveness. Forage crops (both legumes and grasses) are next in effectiveness because of their relatively dense cover. Small grains such as wheat and barley are intermediate and offer considerable obstruction to surface wash. Row crops such as maize, soybeans and potatoes, offer relatively little living cover during the early growth stages and thereby leave the soil susceptible to erosion unless residues from previous crops cover

TABLE 1.21.2 Average annual residue, runoff, soil and nutrient losses and some soil properties for different tillage and crop systems, 1992–98

	PW[a]			WBL[a]		
Measured parameter	MB[b]	DK[b]	NT[b]	MB[b]	DK[b]	NT[b]
No. of runoff events	10	9	8	11	9	9
Residue (%)	9.16	58.6	73.6	7.34	52.7	69.1
Runoff (mm)	96.2	63.4	61.6	107.4	70.8	68.6
Soil loss ($g\,m^{-2}$)	131.8	89.6	77.7	186.1	127.9	109.3
Nitrogen loss ($kg\,ha^{-1}$)	3.65	1.37	1.28	4.73	2.41	1.58
Phosphorus loss ($kg\,ha^{-1}$)	1.02	0.40	0.36	1.68	0.76	0.62
Organic matter ($g\,kg^{-1}$)	12.8	14.2	15.7	10.7	11.7	12.8
Infiltration capacity ($mm\,h^{-1}$)	187	227	252	131	156	178
Hydraulic conductivity ($mm\,h^{-1}$)	262	321	353	184	221	249
Bulk density ($g\,cm^{-1}$)	1.53	1.41	1.32	1.53	1.49	1.44
Aggregate stability[c] (%)	58.2	68.4	75.7	48.6	55.3	62.4

[a]Crop rotation: pasture–wheat (PW) and wheat–barley–grain legume (WBL).
[b]Soil tillage: mouldboard (MB), disking (DK) and no-till (NT).
[c]% of aggregates 2–8 mm which maintained a diameter >1 mm after wet sieving.

the soil surface. Many rangelands lose large amounts of soil under natural conditions, but accelerated erosion can lead to even greater losses if human influences are not carefully managed. Overgrazing, which leads to the deterioration of vegetative cover on rangelands, is a prime example. Grass cover generally protects the soil better than the scattered shrubs that usually replace it under the influence of poorly managed livestock grazing.

There is good experience in Albania of dealing with control measures for gully erosion. If small enough, gullies are filled in, shaped for smooth water flow, sown to grass and thereafter left undisturbed to serve as grassed waterways. When gully erosion is too active to be checked in this manner, more extensive treatment is used. If the gully is still small, a series of check dams about 0.5 m high may be constructed at intervals of 4–10 m, depending on the slope. With very large gullies, permanent dams of concrete or stone are installed in the channel itself.

In forest areas of Albania, most erosion is associated with timber-harvesting practices and forest road construction. For the sake of future forest productivity and current water quality, foresters must become more selective in their harvesting and invest more in proper road construction. On the steepest, most erodible sites, environmental stewardship may require that the timber harvest be foregone and the land be given only protective management. On somewhat less susceptible sites, selective cutting may be practiced without detrimental results. Clear-cutting should be used only on gentle slopes with stable soils.

Poorly built logging roads may lose much soil by erosion of the road surface, the drainage ditch walls or the soil exposed by road cuts into the hillside. Skidding trails that lead runoff water downhill towards a yard area invite the formation of gullies. Repeated trips dragging logs along the same secondary trails also greatly increase the amount of mineral soil exposed to erosive forces. Both practices should be avoided, and yard areas should be located at the highest elevation, most level and well-drained areas available.

Erosion control systems must be developed in collaboration with those who use the land. In Albania, very little land is available to each farmer for food production and many farmers must use all land capable of production simply to stave off starvation and impoverishment. Many examples from Albania and around the world make it clear that when governments cajole, pay or force farmers into installing soil conservation measures on their land, the results are unlikely to be long lasting. Usually, farmers will abandon the unwanted practices as soon as the pressure is off. On the other hand, if scientists and conservationists work with farmers to help them develop and adopt conservation systems that the farmers feel are of benefit to them and their land, then effective and lasting progress can be made. Experience with conservation tillage systems, mulch farming systems and vegetative contour barriers in Albania has shown that farmers can help develop practices that are good for their land and for their profits.

Satisfying as the progress in recent decades is, soil losses by erosion are still much too high. Continued efforts must be made to protect the soil and to hold it in place. The battle to bring erosion under control has only just begun in Albania.

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1.22

Serbia and Montenegro

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1.22.1 NATURAL FEATURES

The Republic of Serbia has an area of 88 361 km^2 characterized by relief varying from extensive lowlands in the north (Autonomous Province of Vojvodina), through hilly terrain and valleys of the south, to mountainous districts in the western, southern and eastern parts of the country. Most of Serbia belongs to the temperate climate belt with rainfall irregularly distributed in time and space. The Republic of Montenegro, with an area of 13 812 km^2, is characterized by predominantly mountainous relief. In this relatively small space, there are great climatic variations: severe mountainous climate in the northeast and mild Mediterranean climate in coastal areas. The relief and rainfall of Serbia and Montenegro offer favourable conditions for water erosion. Serbia and Montenegro are divided into six pedogeographic regions (Resulović *et al*., 1991) (Figure 1.22.1 and Table 1.22.1). Land use in Serbia and Montenegro influences the development of intensive processes of water and wind erosion (Table 1.22.2).

1.22.2 WATER EROSION AND SEDIMENT TRANSPORT IN SERBIA AND MONTENEGRO

Water erosion is the dominant type of erosion in upland regions of Serbia and Montenegro. Water erosion occurs due to rainfall and runoff on sloping land and fluvial erosion in watercourses. The processes of water erosion in Serbia are of different intensities (Table 1.22.3). The commonly used classification of

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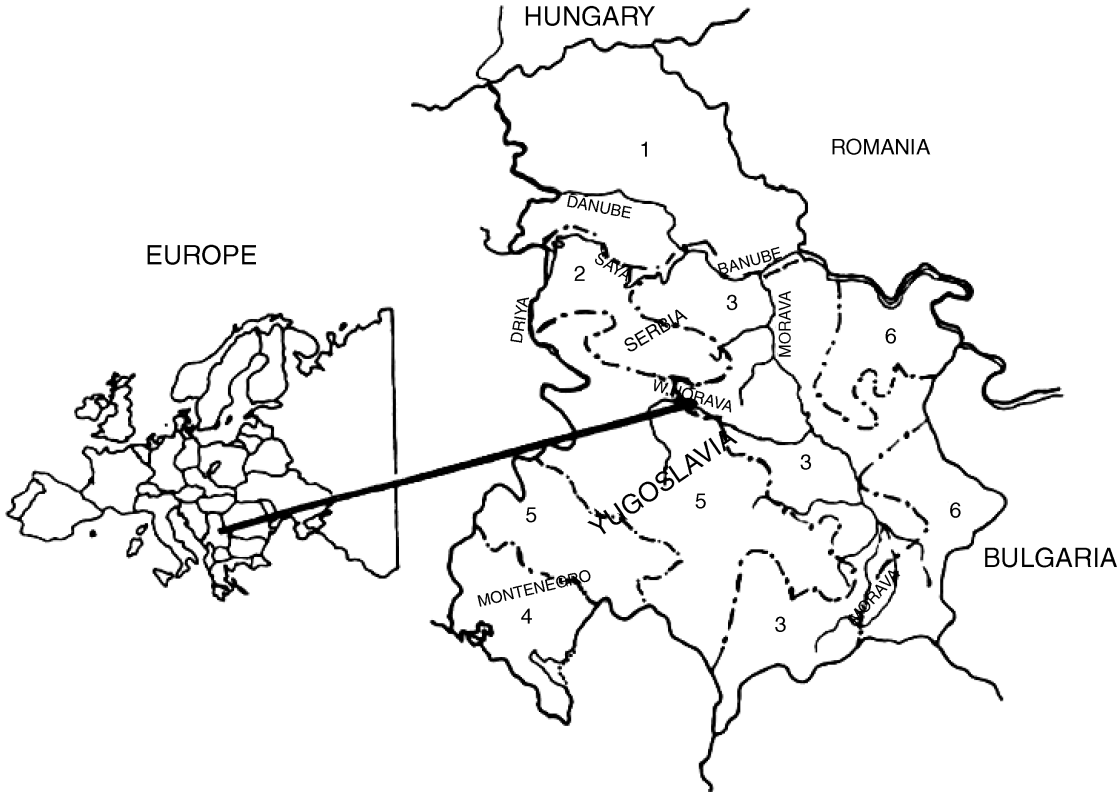

Figure 1.22.1 Pedogeographic regions in Serbia and Montenegro

erosion (Gavrilović, 1972) has five categories (see Figure 1.22.2 and 1.22.3, where Z is the erosion coefficient and W_p is the average annual value of gross erosion for each category of erosion). The Erosion Map (prepared according to the method of Gavrilović, 1972) presents the share of individual categories of water erosion.

Excessive erosion in Serbia is best represented in the catchments of the River Južna Morava and Beli Drim (about 7 % of the total catchment area) and in smaller catchments in the case of Pčinja (18 %). Severe erosion occurs in the catchments of Pčinja (39 %), Lepenac (26 %) and Zapadna Morava (29 %).

Erosion and torrents are an omnipresent occurrence in Montenegro. Depending on the parent rock, there are specific forms of erosion characteristic of karst regions, which are dominant, but there are also other forms of erosion. Excessive erosion in Montenegro occurs mostly in the catchment of the River Ibar (5.2 %) and coastal catchments (4.6 %) and in the smaller catchment of the River Piva (2.8 %). Severe erosion occurs in the catchments of the Rivers Ibar (17.8 %), Zeta (17.8 %) and Cijevna (19.7 %).

Soil losses were measured for short periods on erosion plots (runoff plots) at several localities in Serbia (Đorović, 1990), Sediment transport in small torrential catchments, as a consequence of soil erosion, was measured only in a few experimental catchments (Kostadinov, 1996) in the west and south-east of Serbia. Water erosion causes great damage, such as soil loss, water loss, loss of nutrients, disturbance of runoff regime in the catchments, catastrophic floods and silting of reservoirs.

TABLE 1.22.1 Natural conditions in Serbia and Montenegro

Parameter	Pedogeographic region					
	1	2	3	4	5	6
Average annual temperature (°C)	10.8	11.2	10.0–11.9	15.3	4.7	11.3
Average annual rainfall (mm)	556	678	600–924	4579	779	592
Soil type	Chernozem, Fluvisol, Arenosol, halomorphic soils, district and eutheric Cambisol	Chernozem, eutheric brown soil, Fluvisol, district brown soil	Eutheric brown soil, Vertisol, Chernozem on limited area; Fluvisol in the river valleys	Red soils (terra rosa), eutheric brown soils	Black soil, brown eutheric Calcocambisol and Luvisol, rankers, brown district soil	Eutheric Calcocambisol, terra rosa, district brown soils and Luvisol, alluvial soils
Relief	Loess plateaus and alluvial erraces	Mostly flat	Hilly–mountainous region	Mountainous region and coastal zone	Mountainous relief	Mountainous relief
			Water erosion			
Type of soil erosion	Wind erosion dominant	Sheet, rill	Sheet, rill, gully, landslides, wind erosion (near town of Ram)	Sheet, rill, karst erosion	Sheet, rill, gully, landslides, karst erosion	Sheet, rill, gully, landslides

TABLE 1.22.2 Land use in Serbia and Montenegro

		Ploughed land and vineyards		Meadows, pastures and orchards		Forests		Other	
Republic	Total area (km^2)	km^2	%	km^2	%	km^2	%	km^2	%
Serbia	88,361	37,295	42.2	19,257	21.8	23,592	26.7	8,217	9.3
Montenegro	13,812	645	4.7	4,683	33.9	4,608	33.4	3,876	28.0
Total	102,173	37,940	37.1	23,940	23.4	28,200	27.6	12,093	11.9

1.22.3 WIND EROSION

Wind erosion predominates in the northern part of Serbia, i.e. Autonomous Province of Vojvodina, pedogeographic region 1 (the region of sandy areas Deliblato and Subotica-Horgoš) and on the right side of the Danube river (the region of sandy area Ram Sands). Other lowland terrains are also exposed to occasional severe action of winds, causing serious erosion. Wind erosion causes great damage such as soil loss covering the most productive soils with poor-quality wind sediments and loss of seeds, artificial fertilizers and plant protecting agents. Soil erosion (by water and wind) and sediment transport, in addition to other damage, also cause environmental damage (soil loss, water loss, mechanical and chemical pollution of water, degradation of the landscape, etc.) (Kostadinov *et al*., 1997).

The measures to reclaim the situation are complex and costly. In 1981, near the village Tavankut (Subotica-Horgoš Sands, northern part of the Province of Vojvodina), a stormy north-west wind removed about 194 t ha^{-1} of sandy soil from a newly established orchard in one day (Letić *et al*., 1995). Calculated by Pasak's method, the average intensity of wind erosion for the province of Vojvodina is 1.82 t ha^{-1} yr^{-1}(Pasak, 1985).

1.22.4 CONCLUSIONS

Soil erosion (by water and wind) is the most serious form of soil degradation in Serbia and Montenegro. Practically all of the country is affected by water erosion (wind erosion occurs in the Province of Vojvodina).

TABLE 1.22.3 Distribution of water erosion intensity in Serbia and Montenegro

		Republic			
		Serbia		Montenegro	
Category	Erosion processes intensity	km^2	%	km^2	%
I	Excessive erosion	2,888	3.27	180	1.3
II	Intensive erosion	9,138	10.34	1,354	9.8
III	Medium erosion	19,386	21.94	5,027	36.4
IV	Weak erosion	43,914	49.78	6,367	46.1
V	Very weak erosion	13,035	14.75	884	6.4
Total		88,361	100	13,812	100

Source: Water Resources Management Basic Plan of Serbia (1996) and Water Resources Management Basic Plan of Montenegro (2001).

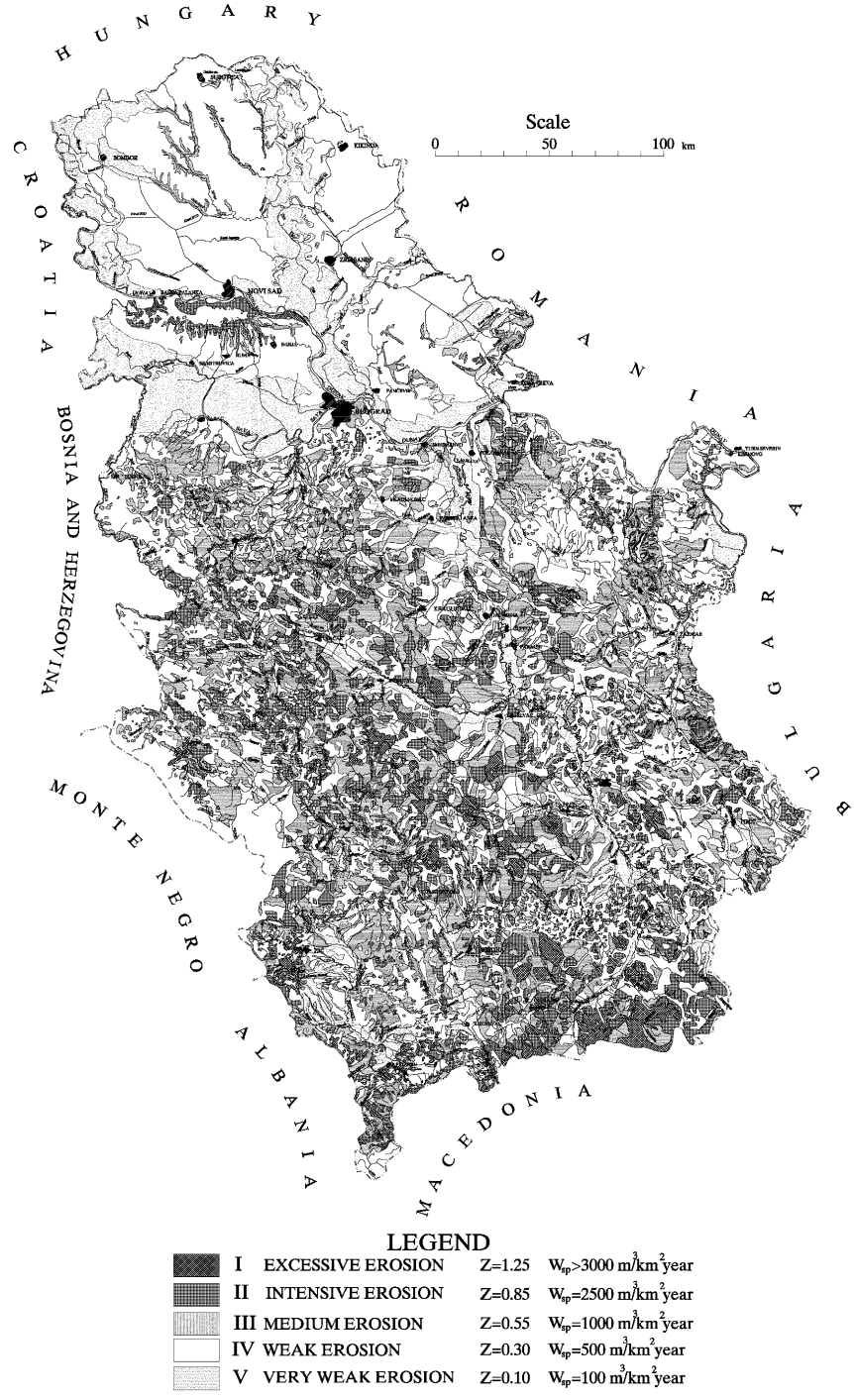

Figure 1.22.2 Erosion map of Serbia. *Source*: Water Resources Management Basic Plan of Serbia (1996)

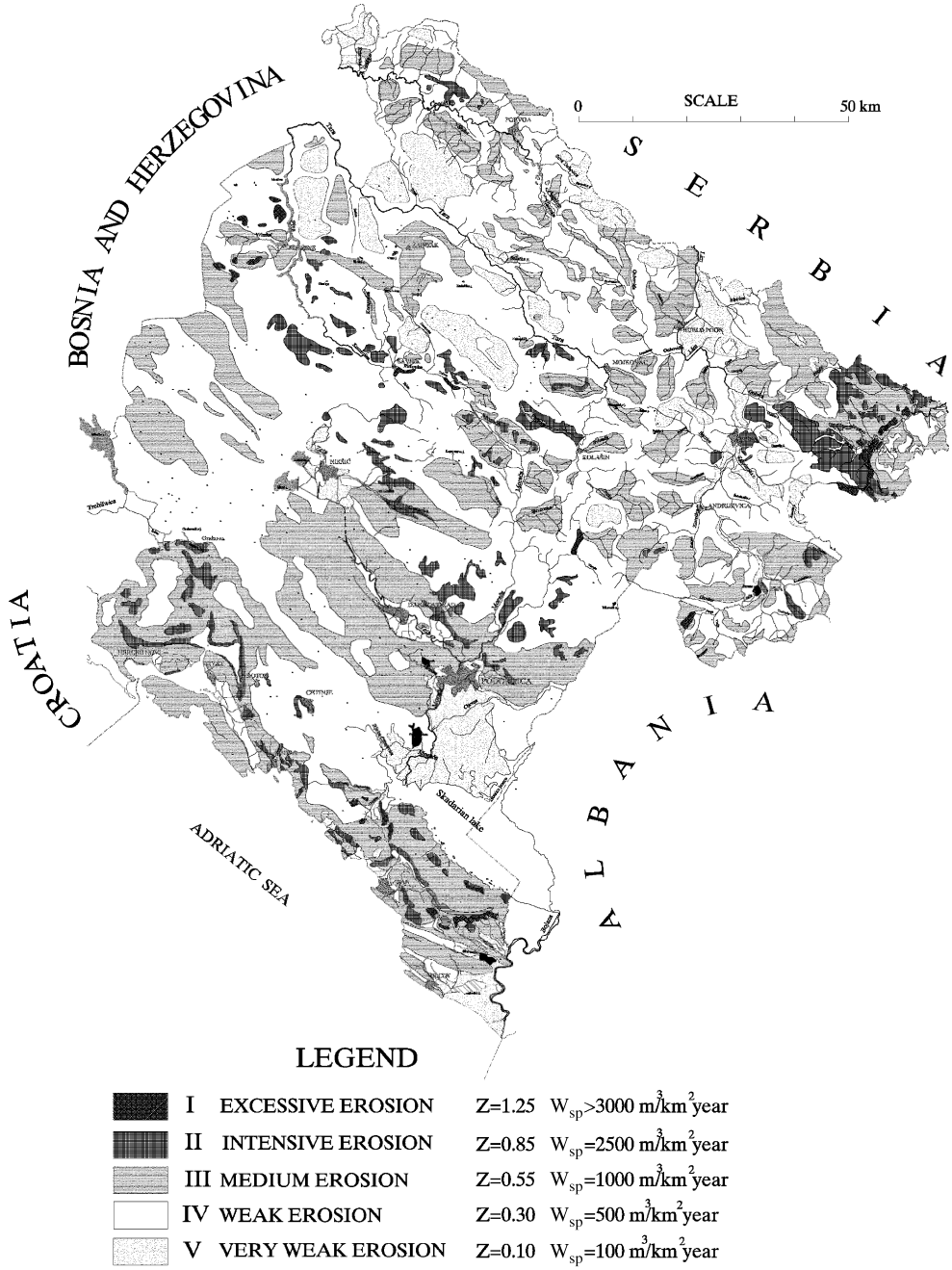

Figure 1.22.3 Erosion map of Montenegro. *Sources*: Water Resources Management Basic Plan of Montenegro (2001)

TABLE 1.22.4 Gross erosion and sediment transport in Serbia and Montenegro

Republic	Gross erosion		Annual sediment transport	
	Total ($m^3\ yr^{-1}$)	Specific ($m^3\ km^2\ yr^{-1}$)	Total ($m^3\ yr^{-1}$)	Specific ($m^3\ km^2\ yr^{-1}$)
Serbia	37,249,975	421.57	9,350,765	105.80
Montenegro	17,306,436	1,253.00	4,857,253	351.67
Yugoslavia	54,556,411	533.96	14,208,018	139.06

Source: calculated by the method of Gavrilović (1972).

Control of soil erosion is the first priority for the prevention of soil degradation in Serbia and Montenegro. In Serbia and Montenegro, relatively good results in erosion and torrent control have been achieved since the beginning of erosion control work (ECW) in 1907. Also, good results have been achieved in the reduction of sediment transport, but as some dams were built before ECW they silted up with eroded sediment. The entire period since the beginning of ECW has been characterized by the absence of the necessary understanding by the State, which is reflected in its investments in ECW. Sometimes good legal solutions (concerning erosion control) were adopted but were not strictly obeyed in practice because of the absence of penalties. The constant lack of finance has been a major problem.

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1.23

Greece

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1.23.1 PHYSICAL GEOGRAPHY

Greece is a rugged, mountainous country with great variation in altitude in relatively short distances. About 49% of the surface area is characterized by slopes greater than 10%, whereas only 36% comprises lowlands with slopes less than 5%. Areas higher than 800 m occupy 28.6% of the country, with Mount Olympus being the highest peak (2917 m). Owing to its predominantly steep terrain and adverse bio-climatic conditions, the country faces considerable soil erosion problems.

Owing to their steep slopes, the soils on the mountains are extremely eroded, shallow, rather poor and classified as Lithosols (FAO–UNESCO, 1989), and are useless for any agricultural activity. On the uplands, the steep slopes combined with the destruction of natural vegetation (fire, cultivation, overgrazing) have caused soil erosion and formation of Cambisols, Luvisols and Regosols, or Lithosols if the parent rock is exposed at the surface. The soils on the lowlands are far more productive. They can be subdivided into two main groups: (a) soils formed after the reclamation of former lakes and (b) soils formed on alluvial plains, moderately to highly fertile. They are mainly classified as Fluvisols, Cambisols, Luvisols and Vertisols.

Based on soil, climatic and topographic characteristics, land of potentially high quality covers 19% (24 919 km^2) of the total land surface (CORINE, 1992), land of moderate quality 18% or 23 394 km^2 and land of low quality 57% or 75 775 km^2. The last type of land is mostly used for traditional, low-capital intensity farming systems, which are important for maintaining the characteristic Mediterranean landscape.

The climate of Greece is of Mediterranean type, with most rains falling in the cool period, from October to March, whereas the hottest months, July and August, are almost without precipitation. The amount of rainfall ranges from 780 to 1280 mm yr^{-1} in the western part of Greece, this amount being approximately halved in the

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TABLE 1.23.1 Changes in the main land use categories in Greece, 1971–2001 (area in thousands of hectares)

Year	Cultivated	Pastures	Forests	Horticulture	Orchards	Vineyard	Fallow	Total
1971	2532.9	—	2967.5	114.3	698.9	219.8	—	6533.4
1981	2423.8	5252.2	2951.1	122.1	840.0	186.4	506.4	12282.0
1991	2334.4	—	2937.8	123.6	924.6	151.7	—	6472.1
2001	2213.2	5219.1	—	118.9	997.7	134.3	454.5	9137.7

Source: National Statistical Service of Greece, 2002.

eastern part, ranging from 380 to 640 mm yr^{-1}. Potential evapotranspiration exceeds rainfall for more than 6 months, creating large water deficits for the growing plants in the period from May to October.

Most rains are of high erosivity, unevenly distributed during the year and may interrupt appreciably long dry spells. The major part of Greece is characterized by intense storms of short duration, which occur in the dry season. Rain erosivity is highest especially in the western and south-western parts of the Greek mainland, and also in the eastern Aegean islands and for the greater part of Crete. Lower erosivities occur in the north-western parts, and the medium erosivity class applies mostly to the central and eastern parts of the mainland (Yassoglou and Kosmas, 1988).

Natural vegetation in Greece is typical of the dry climatic conditions of the eastern Mediterranean. As Table 1.23.1 shows, no significant changes have occurred in recent decades in the various land-use types. The most important changes are observed in orchard areas.

1.23.2 HISTORICAL EVIDENCE FOR SOIL EROSION

Greece may serve as a typical semi-arid Mediterranean country having been largely affected by soil erosion, and facing severe problems of desertification. Its hillsides, originally forested, were covered (6000 BC) by a fertile but shallow soil vulnerable to erosion. Great deforestation took place for fuel, house building, ship construction and other purposes (4300 BC). Shepherds damaged the forests with fire to eradicate the woody vegetation and encourage the growth of grass, which was then overgrazed. Farmers contributed to the damage by growing cereals, olives, figs, grapes and other fruits and vegetables on sloping ground with insufficient soil protection.

Upland grazing and then farming in Greece probably began around in the middle of the second millennium BC, and it was greatly intensified during the Hellenic period (800 BC). As the soil eroded, the Greeks had to shift from food grains to commercial crops of grapes and olives, which could be grown on thinner soils. By the time of the Macedonian hegemony in 338 BC, the land had already deteriorated markedly. During the later Hellenistic period, efforts were made to expand the agricultural land of Greece by draining marshlands.

Critias mentions that in one of his dialogues Plato states (427–347 BC): 'What now remains on the formerly rich land is like the skeleton of a sick man, with all the fat and soft earth having worked away and only the bare framework remaining. Formerly, many of the mountains were arable. The plains that were full of rich soil are now marshes. Hills that were once covered with forests and produced abundant pasture now produce only food for bees'.

The term land degradation has been found in the code of the Emperor Theodosius (438 AD) with numerous references to *agri deserti* or regions abandoned owing to their low productivity or as a consequence of military campaigns. Copper and Bronze Age remains show that most of the cultivated areas in the islands of the archipelago were being exploited by the second millennium BC. Farming with vines, olives, and cereals had

been established on the islands and particularly in Crete. Pressure on land resources resulted in the construction of agricultural terraces in Minoan Crete. A number of artificial terraces to protect against soil erosion on Delos, an economic centre of Greek antiquity, must have been constructed before 400 BC. Many hillside terraces on Crete which are dated to the 16th century were used for vines and cereal cultivation, the latter being displaced by vineyards (Grove, 1996).

1.23.3 CURRENT EROSION PROCESSES

Soil erosion in Greece has proceeded at high rates in the last 50 years following the intensification and mechanization of cultivation. One of the most spectacular examples of severe soil erosion is the complete removal of the thick, dark, surface soil horizon that occurred in the hilly Tertiary landscapes of central Greece at rates exceeding 1 cm yr^{-1} (Danalatos, 1993). Erosion processes mainly responsible for land degradation in Greece are related to water, wind and tillage erosion.

1.23.3.1 Water Erosion

Water erosion constitutes a major problem of land degradation found in the hilly areas of Greece (Figure 1.23.1). Soil erosion is attributed to climatic conditions, vegetation cover and land-use management practices. The large-scale deforestation of hilly areas occurring in recent decades, accompanied by intense cultivation and overgrazing, resulted in accelerated erosion and the formation of badlands with very shallow soils.

Extensive eroded areas are confined to rock formations primary the Mesozoic limestone and secondarily acidic igneous and metamorphic rocks. Drier microclimatic conditions prevail on these areas, reducing the potential for plant growth, and the soils remain bare for long periods, favouring overland flow and erosion. The soils on these areas are very shallow or the parent rock is exposed at the surface.

Soil textures on acidic rocks are usually moderately coarse to medium and the soil aggregate stability is low. Therefore, soil erodibility on these rocks is high and soil erosion has affected large areas. It is estimated that about 28% of the country's land surface is partly degraded owing to the presence of such parent materials.

Hilly areas with a substratum of shales/sandstones or flysch exhibit a lower erosion risk. The soils are moderate- to fine-textured and permeable, and have a moderate to rich vegetation cover. However, if the natural vegetation is removed (by fires, forest clearance, etc.), the areas on flysch become very susceptible to gully erosion and the occurrence of landslides.

Erosion measured on various types of land use, such as cereals, vines, olives, bare land (Table 1.23.2), and shrubby vegetation, depends considerably on the type of vegetation. Rainfed cereals cover a large part of the country's uplands. The most crucial period for soil erosion under rainfed cereals is from early October to late February, when the soils are almost bare or partially covered by the growing crop. Erosion rates measured in hilly areas located in Thiva (central Greece) and Petralona (northern Greece) ranged from 0 to 52 t km^{-2} yr^{-1} (Kosmas *et al.*, 1997; Stamou, 1995). Today, production of rainfed cereals in the hilly areas has declined owing to high degradation of the soils, and they are now used as pastures.

Perennial crops such as almonds and vines occupy extensive hilly areas, even though vines have declined during recent decades. These areas require frequent removal of annual vegetation using pesticides (weed control) or ploughing the soil. Such soils remain almost bare during the whole year, whereas the frequent use of heavy machinery negatively affects aggregate stability and organic matter content, creating favourable conditions for overland flow and soil erosion. Soil erosion rates measured in vineyards in the Attica area ranged from 15 to 252 t km^{-2} yr^{-1} (Kosmas *et al.*, 1997).

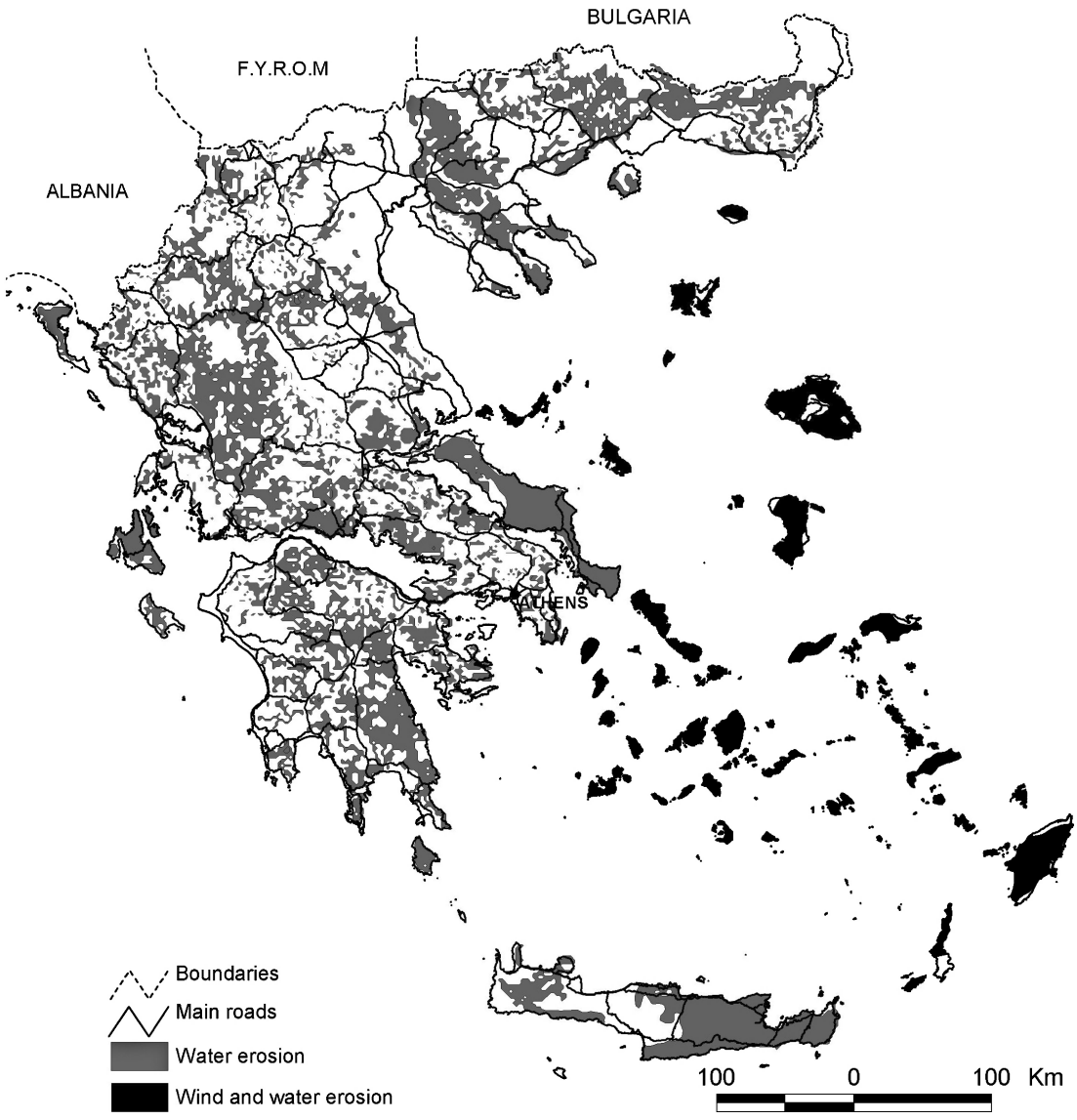

Figure 1.23.1 Hot spots in Greece for water and wind erosion

The lowest rates of runoff and sediment loss usually occur in olive groves under semi-natural conditions and certain type of management such as maintaining vegetation of annual plants, which in combination with the dense leaf canopy of the trees efficiently protect the soil surface from raindrop impact. Under such conditions, water runoff and sediment loss are negligible. Soil erosion data measured in the Attica area and Zakynthos island have shown that erosion may range from 0 to 6 t km^{-2} yr^{-1}.

On shrublands which are used mainly for grazing, soil erosion rates depend on annual rainfall, vegetation cover and livestock density. Erosion rates measured in Zakynthos ranged from 0.2 to 1.6 t km^{-2} yr^{-1}, whereas higher erosion rates are expected in the eastern part of the country (Kosmas *et al.*, 1997). Soil depth and type of

TABLE 1.23.2 Soil erosion rates measured on runoff plots with calcic xerochrept soils for different land use types (R = annual rainfall in mm, SL = soil loss in $t\,ha^{-1}\,yr^{-1}$, plot size = 10 × 3 m and slope gradient = 7–23%): Spata experimental field site (number of plots = 35)

Year	1991		1992		1993		1994		1995		1996		1997	
	R	SL	R	SL	R	SL	R	SL	R	SL	R	SL	R	SL
Olives		2.8		0		0		0,1		0		0.2		0.3
Vines	472	10.4	339	0	402	1.2	799	19.1	292	11.9	302	24.5	307	4.2

parent material affect vegetation cover and erosion rates in shrublands. Long-term studies conducted in various shrublands or abandoned lands of Greece showed that two classes of soil depth may be distinguished for land protection: (a) a critical soil depth (25–30 cm), below which the recovery of the natural vegetation is very low, and the erosional processes may be very active resulting in accelerated soil erosion rates, and (b) a crucial soil depth (4–10 cm) under which perennial vegetation cannot be sustained and the soil is rapidly removed by wind or water erosion, so that degradation of the land is an irreversible process (Kosmas *et al.*, 2000).

1.23.3.2 Tillage Erosion

Tillage erosion is considered one of the most important processes of land degradation in hilly cultivated areas in Greece. Extensive areas have largely degraded during recent decades owing to erosion caused by the use of heavy, powerful tillage implements. The availability of heavy, powerful machinery favoured deep soil ploughing at high speeds in directions usually perpendicular to the contours. This resulted in the displacement of huge amounts of soil materials from the upper convex hillslope parts (summit, shoulder, backslope) to the lower concave parts (footslope, toeslope), significantly decreasing the productivity of the convex positions, especially when subsurface limiting soil layers occur, such as petrocalcic horizons or bedrock. It is estimated that 8% of the hilly agricultural land in Greece has been abandoned in recent decades owing to the diminished productivity caused by soil erosion (Kosmas, 1999).

Tillage erosion exposes subsoil, which may be highly erodible by wind or water, and fills in ephemeral flow areas, acting as a delivery mechanism for water erosion. Studies based on (a) the comparison of preserved areas with neighbouring cultivated ones in Thessaly (central Greece), (b) previous records of soil characteristics in certain areas (Papoutsopoulos and Tzorykin, 1936) and (c) soil erosion data obtained under agricultural management practices and similar land conditions (Tsara *et al.*, 2001) clearly demonstrate that tillage rather than water erosion is the most important factor controlling land degradation in hilly cultivated areas. Water erosion in areas cultivated with cereals, vines or olives is responsible for a loss of a few millimetres (1–3 mm) of soil per year or even less (Kosmas *et al.*, 1997). The estimated total annual soil loss in the same areas cultivated mainly with cereals reaches 4–14 mm per year (Kosmas *et al.*, 2000). Soil studies in Thessaly hilly areas showed that soil depth has been reduced by 24–30 cm in the period from 1936 to 1999.

Measurements conducted in hilly areas in Greece (Atica, Thiva, Thessaly) to assess tillage erosion under existing management practices (Gerontidis *et al.*, 2001; Kosmas *et al.*, 2001) show that soil displacement is greatly affected by plough depth, tillage direction and slope gradient. On steep hillslopes (22 % slope gradient), a maximum soil displacement of 97 cm was measured in the plough layer after ploughing the soil down slope to a depth of 40 cm and perpendicular to the contour lines (Figure 1.23.2). Under the same soil conditions and management practice, soil displacement was reduced to 69 cm after ploughing the soil along the contour lines (Figure 1.23.2). A 50 % reduction in plough depth can reduce soil displacement by more than 75%. The up-slope reversion of furrows with the tractor moving parallel or perpendicular to the contour lines significantly reduces soil displacement, ranging from 2 to 33 cm.

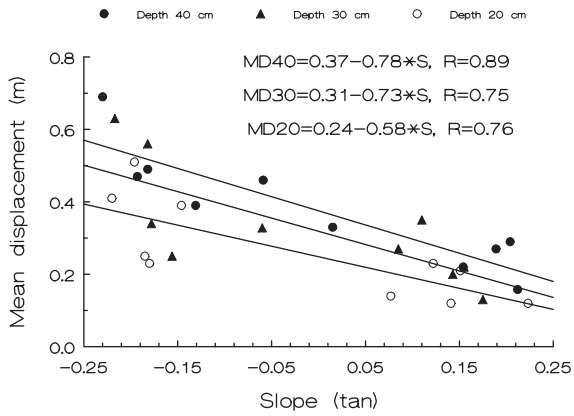

Figure 1.23.2 Relation of mean soil displacement and slope gradient for three plough depths with tillage direction perpendicular to the contours. (Reprinted from *Soil and Water Conservation Journal* **56**, Gerontidis *et al*., The effect of mouldboard plough on tillage erosion along a hillslope, pp. 147–152, 2001, with permission from Soil and Water Conservation Society)

1.23.3.3 Wind Erosion

Wind erosion is another process of soil erosion that occurs especially in the semi-arid part of Greece. However, information on the extent of wind erosion in Greece is limited. Areas more vulnerable to wind erosion are the islands of the Aegean Sea (Figure 1.23.1) and the north-eastern part of the mainland. Strong north or north-eastern winds prevail during the dry period in Greece, creating favourable conditions for wind erosion. The main factors controlling wind erosion in Greece are vegetation cover, slope exposure, soil water deficit, grazing and fires. Mainly steep slopes with shallow soils and semi-arid climatic conditions characterize the Aegean islands. The vegetation cover may range from bare to fully covered depending on slope gradient, slope exposure, soil depth, parent material and grazing intensity. Under dry climatic conditions, perennial vegetation cannot grow and only annual vegetation is present during the wet period. If the land is grazed, soils are partially covered during the summer period, favouring conditions for wind erosion. Fires destroy the existing vegetative cover and contribute to wind erosion by exposing the soil surface to wind action.

Slope exposure affects wind erosion. The highest wind erosion rates in Greece are expected on northern- and north-eastern-facing slopes. Winds blow from southerly directions normally during the wet period, when the soils are relatively wet and protected by natural vegetative cover. Therefore, southern and south-western exposures are less affected by wind erosion.

A soil water deficit, occurring during summer and early autumn, creates favourable conditions for soil particle detachment and wind erosion. Animal pathways are also vulnerable to wind erosion. Animal trampling on certain pathways destroys soil aggregates leaving a layer of dust easily suspended in the air.

Practical measures for reducing wind erosion are limited. In recent decades, extensive abandonment of agricultural land on the Aegean islands owing to diminished productivity or the shifting of land use from farming to grazing resulted in more effective soil protection against wind erosion, due to the growth of natural vegetation.

1.23.4 MAJOR ON- AND OFF-SITE PROBLEMS AND COSTS

The effects of soil erosion caused by surface water runoff, tillage or wind are either on- or off-site. On-site effects are those caused by various erosion processes where erosion occurs. Off-site effects include any others, initially generated by erosion processes, but at greater distances.

1.23.4.1 On-site Effects

These can be either long-term such as the progressive degradation of soil or short- to medium-term such as the effect on crop production. The main on-site effects in Greece are the removal of fertile topsoil, removal of organic matter, leaching of plant nutrients, exposure of large amounts of rock fragments or bedrock at the surface, reduction of crop yields, desertification and abandonment of the land. Soil particles are not equally removed by water and wind. Small soil particles are preferentially washed out. Hence the general effect is a gradual increase in sand particles. The change in soil texture leads to a significant reduction in crop production due to the loss of nutrients (which are associated with clay particles) and a decrease in water holding capacity. Furthermore, soils on hilly areas are mainly shallow owing to their formation on consolidated parent materials. Any further loss of soil drastically reduces the effective rooting depth, leading to a sharp reduction in crop production.

Analytical data on soil organic matter content of the topsoil of hilly lands in Thessaly have shown a reduction from 2.6 to 1.5 % during the last six decades. This is mainly attributed to management practices (oxidation, burning of the remaining plant residues) and losses due to water erosion. The removal of organic matter reduces soil aggregate stability, adversely affecting soil erosion.

Soil erosion caused by water can remove large amounts of N- and P-fertilizers, especially when applied to the soil surface. Traditionally, fertilization of vines, olives, cereals and almonds is achieved by spreading mixed fertilizers on the soil surface during the winter period. Under such conditions, large amounts of nutrients can be transported to surface waters, causing pollution problems and eutrophication of lakes and reservoirs. Measurements of nitrogen loss by runoff water, which have been conducted in vineyards in the Attica area, have shown losses up to 9.5 kg N ha^{-1} yr^{-1}.

Redistribution of topsoil from upper landscape positions by the various tillage operations reduces significantly the effective soil depth and the water holding capacity, which is the most serious loss in the long-term, restricting productivity (Danalatos, 1993; Kosmas *et al*., 1993). Studies conducted in various hilly areas have shown a drastic decrease in cereal production in the upper hillslope positions accompanied by a slight increase in productivity in the lower hillsope positions. As Figure 1.23.3 shows, a reduction of about 26% in biomass production was estimated for an upper hillslope soil over 63 years (1936–99), corresponding to a loss of about 30 cm of soil during that period. An increase of about 14.5 % in wheat biomass production in the lower concave slopes occurred, being much less than the loss on the higher convex slopes (about 26 %).

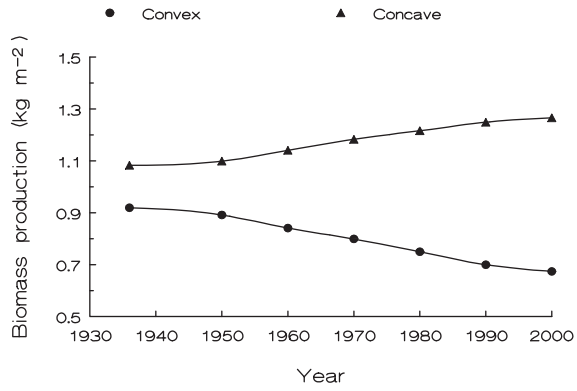

Figure 1.23.3 Estimated average change in wheat biomass production due to erosion in convex and concave slope positions. (Reprinted from *Journal of Soil Use and Management* **17**, Tsara *et al*., The long-term effect of tillage on soil displacement of hilly areas used for growing wheat in Greece, pp. 113–120, 2001, with permission from British Soil Science Society)

According to the Greek Action Plan to combat desertification, 30% of the territory faces severe problems of desertification. The land has highly degraded mainly owing to soil erosion and, under dry climatic conditions, vegetation growth and crop production have declined substantially in those areas. Cultivation of highly degraded land is not economically feasible, leading to land abandonment with adverse consequences for the local economy.

Forest fires are one of the most important factors in soil erosion in Greece. Fires have become very frequent in the pine-dominated forests with an average rate of 52 000 ha yr^{-1} in the last decade. Burned forests in dry areas with shallow soils are not easily regenerated and therefore they slowly disappear.

1.23.4.2 Off-site Effects

These can be either short- or medium- to long-term such as damage to crops and infrastructure from uncontrolled runoff and flooding, silting of channels and reservoirs, environmental alterations of wetlands, lakes and estuaries, decline of the economy of local communities and migration of local people. Extensive agricultural lowlands are frequently flooded by water, which destroys the growing crops. Flooding especially during the autumn (fall) or early winter months has often caused extensive damage to roads, houses, cars and stored products.

Silting of water channels is especially important in watersheds having steep slopes and poor vegetation cover. For example, the rivers Sperchios (central Greece) and Evrotas (southern Greece) transport huge amount of sediment every year, deposited in crossing alluvial plains. Torrential creeks originated from watersheds with low vegetation cover and shallow soils formed mainly on igneous rocks are filled up every year with huge amounts of boulders, coarse gravel and sand, aggravating the flooding problems. Fourteen dams have been constructed in various places in Greece with a water storage capacity of 9551×10^6 m^3. Measurements conducted in western Greece where flysch formations are predominant, and in a number of sites in eastern Greece showed that the sediment load transported to dams ranged between 1200 and 2000 t km^{-2} yr^{-1} (Zarris et al., 2002).

Transportation of sediments to wetlands, lakes, estuaries, etc., alters the natural ecosystems. The National Park of Prespa wetland protected by the Ramsar Treaty receives large amounts of sediments, reducing the area of land in which unique bird species can nest. Aerial photo-interpretation has shown that the land has expanded into Prespa Lake by about 30 m in 35 years. The coastline of the National Park of Zakynthos island receives large amounts of fine particles by the floods, adversely affecting the nesting of sea turtles along the coast.

Soil erosion and land degradation of hilly areas affect crop production and therefore farmers' income, resulting in massive migration, especially of younger people, to the cities. Obviously, this migratory flux is accompanied by significant demographic decline of the countryside. The decline in local economies and rising unemployment levels are now faced by the European Union with its economic support for agricultural areas in an effort to maintain rural populations in environmentally sensitive areas so as to ensure their proper management.

1.23.5 SOIL CONSERVATION AND POLICIES TO COMBAT EROSION AND OFF-SITE PROBLEMS

The Greek Government undertakes soil conservation and protection measures against soil erosion, but these measures are focused mainly on forested areas. At a national level these areas are under the control of the Department of Forestry and Natural Environment of the Ministry of Agriculture. The most important actions for protecting hilly areas from erosion are reforestation of areas recently affected by fire

and the construction of permanent conservation structures. Reforestation of such areas is relatively low with an average annual rate of less than 10% of the burned areas. The purposes of conservation structures such as reinforced concrete, drop spillways and drop inlets are the conservation of water and protection from erosion.

Many uplands have been terraced for cultivating cereals, vines, olives and other crops. In many cases, terraces which have been constructed with stones, are hundreds or even thousands of years old. Recently, most of these areas have been abandoned. As a result, many of the existing terraces have collapsed, and this causes a rapid removal of the soil by runoff water. Considering that such terraces protect very valuable soil for preserving the existing vegetation, a pilot programme was initiated in 2002 for protecting terraced land by subsidizing the repair of existing terraces.

Generally, Greek laws for environmental protection are affected by the European Common Agricultural Policy. According to the Commission's Regulation 797/85, Article 19, Member States are explicitly permitted to introduce their own national aid schemes for supporting farming practices that preserve or improve the environment. An annual payment per hectare could be given to farmers in sensitive areas who agree to adhere to appropriate farming methods for protecting the environment. Such practices might involve restricting the intensity of livestock farming, limiting the use of water for irrigation or taking measures to reduce soil erosion, for example by the conversion of arable land to wildlife reserves.

The Regulation 1760/87 issued by the European Commission encourages Member States to define areas of sympathetic agricultural practices. They were able to claim one-quarter of the cost of such schemes up to a limit of €100 ha^{-1}. A total of €319 million was allocated to protection from soil erosion, to biotope management and selective reforestation. In the decade 1982–92, the strategy was progressively modified to allow greater support for conservation of the landscape and its component parts.

The Maastricht Treaty (1992) recognized that the European Union must promote measures at an international level to deal with environmental problems and ensure 'sustainable growth respecting the environment' (Corrie, 1991; Baldock and Beaufoy, 1992a,b). Some 15% of cultivated land was to be put into fallow (set aside). Extending the fallow land could mean an improvement of soil fertility (International Workshop on Land Use Changes and Cover in the Mediterranean Region, 2001).

The 1992 United Nations Convention for Combating Desertification was ratified by the Greek Parliament in 1997 and has become a State law. Based on this convention, the Greek National Plan for Combating Desertification was produced, including guidelines on how to protect land from various processes, including soil erosion. Any development programme related to natural resources has to take into account the measures included in the action plan.

The European Union regulation 1257/99 is financing agro-ecological projects related to the protection of the environment. A project on protection of terraces in some of the Aegean islands was initiated in 2002.

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1.24

Macedonia

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1.24.1 PHYSICAL GEOGRAPHY

The Republic of Macedonia (RM) is located in the central part of the Balkan Peninsula. It is a landlocked country having an area of 25 713 km^2 with approximately 80% of the entire territory in hilly and mountainous regions. About 2 % of the land area is covered by water comprising 35 large and small rivers, three natural lakes (Ohrid Lake, Prespa Lake, Dojran Lake) and over 100 reservoirs. The population of the country is around 2 million, of which about 60 % live in urban areas and the overall population density is 81 inhabitants per km^2. The major urban centres are Skopje, Bitola, Tetovo, Kumanovo, Veles, Prilep, Stip, Ohrid, Strimica and Gostivar.

Industry is the dominant sector, accounting for 35% of the gross social product (GSP) and 39.9% of employment. Agriculture combined with forestry and fishing and the service sector account for 22% and 30% of the GSP, respectively.

Although the RM is small in area, it shows a great diversity of relief forms, geological formations, climate, plants and soils.

The difference in altitude is from 40 to 2764 m above sea level. The territory of the RM belongs to three basins: the Black Sea (44 km^2 or 0.17%), the Adriatic (3359 km^2 or 13.07%) and the Aegean (22 310 km^2 or 86.76%). The main river is the Vardar with a catchment area of 20 545 km^2 (79.9% of the whole territory).

As a result of the heterogeneity of natural conditions, the territory of the RM can be distributed into eight climate–soil–vegetation zones. About 56% of the territory belongs to two zones (continental submediterranean and warm continental). The average annual temperature is 10.5 °C, with absolute extremes of 44.3 and −31.5 °C. Average annual precipitation is 660 mm.

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About 25% (634 000 ha) of the total land area is pasture, 25% (550 000 ha) arable, 55 300 ha meadows, 57 000 ha vineyards and orchards, 8% (20 5000 ha) barren, 37.5% (965 650 ha) forest, 2% lakes and 2.5% urban or industrial land. The 1982 Physical Plan projected an increase in forest land and a decrease in agricultural land over the forthcoming decades. About 60% of the population live in urban areas and over the last 20 years there have been absolute decreases in population in many of the rural areas.

Fertile land is scarce, with 82% of arable land in fertility classes IV–VII. Because of recent declines in the rural population and economic activity, fallow and uncultivated land is increasing in area, comprising about 160 000 ha in 1993, or 30% of arable land. About 70% of arable land is privately owned and plans are under way to privatize the remainder.

Pasture constitutes about 634 000 ha but yields are well below potential, averaging only about 270 kg ha^{-1} (potential yields could be as high as 800 kg ha^{-1}). The low yields may also indicate that some of this land may be more appropriate as forest. In the past, much pastureland in Macedonia, as elsewhere in the Balkans, was previously forested. Pasture is managed by the public enterprise 'Macedonian Pastures'.

Forest reserves cover about 1×10^6 ha or 37.5% of the land area of Macedonia. This is characteristic of oak stands. About 50% of forests comprise pure and mixed oak stands (480 000 ha), 29% (285 000 ha) beech, 8% (80 000 ha) black pine and Scotch pine and 15% other stands. About 67% of forests are coppiced. Degraded forests and shrubs cover 262 000 ha (27%) of the forest land. A substantial proportion of the forest is located on steeply sloping land, where forest cover is necessary for soil conservation and watershed protection purposes, and where logging is restricted.

Urban growth has not always been accompanied by adequate infrastructure development, and urban expansion has frequently taken place on high-quality agricultural land. According to some estimates, about 0.5% of agricultural land is lost annually to construction.

1.24.2 HISTORICAL EVIDENCE FOR EROSION

A lot of natural conditions in RM (climate condition, topography, vegetation cover, geology) contribute to high rates of erosion. Also, poor arable farming, grazing management and deforestation in the past have contributed to erosion, a problem affecting all of the country.

Deforestation was extreme before World War II. In the 13–14th centuries, German miners arrived in Macedonia and started mining activities especially in the east and north-east. They cut forests and used wood for fuel and mining construction.

The Turkish Ottoman Empire governed Macedonia for five centuries from the 14th to the beginning of the 20th century. Wood from forests in the central part of Macedonia was fully cut and transported along the River Vardar to Thessalonica. Today in that area, part of the left side of the river in Central Macedonia is possibly the only semi-desert in Europe.

Forests around the settlements were also degraded. Low education levels, insufficient awareness, social structure and poverty were reasons for extensive forest destruction around the settlements.

1.24.3 CURRENT EROSION PROCESSES

There is an Erosion Map prepared at a scale of 1:50 000 in a database version. This version was finished in 1992. The digital version was finished in 2002.

The empirical methodology of Gavrilovic (which is similar to methodology of Poljakov) was used for mapping erosion intensity.

TABLE 1.24.1 Erosion distribution in RM

	Degradation category (erosion processes)	Area (km^2)	Area (%)	Mean annual erosion ($m^3\,km^{-2}\,yr^{-1}$)
I	Extremely high	698	2.77	>3000
II	High	1832	7.38	1500–3000
III	Medium	6893	27.78	1000–1500
IV	low	7936	31.98	500–1000
V	Very low	7463	30.09	70–500

Of the total area, 96% is affected by processes of erosion. An amount 9423 km^2 or 36.65% of the total state territory is in the highest categories (I–III: Table 1.24.1).

The total annual production of erosive material is about $17 \times 10^6\,m^3\,y^{-1}$ or $685\,m^3\,km^{-2}\,yr^{-1}$, of which $7.5 \times 10^6\,m^3\,yr^{-1}$ or $303\,m^3\,km^{-2}\,yr^{-1}$ is carried away. Significant parts of these deposits, about $3 \times 10^6\,m^3\,yr^{-1}$, is not carried through the downstream sections of the rivers to the exit of the state territory, but sedimented in natural lakes and reservoirs.

Annual soil loss represents an annual average loss of an arable soil layer 2 cm deep over an area of 8500 ha. The economic cost of erosional impacts therefore considerable.

The distribution of erosional processes is shown in Figure 1.24.1

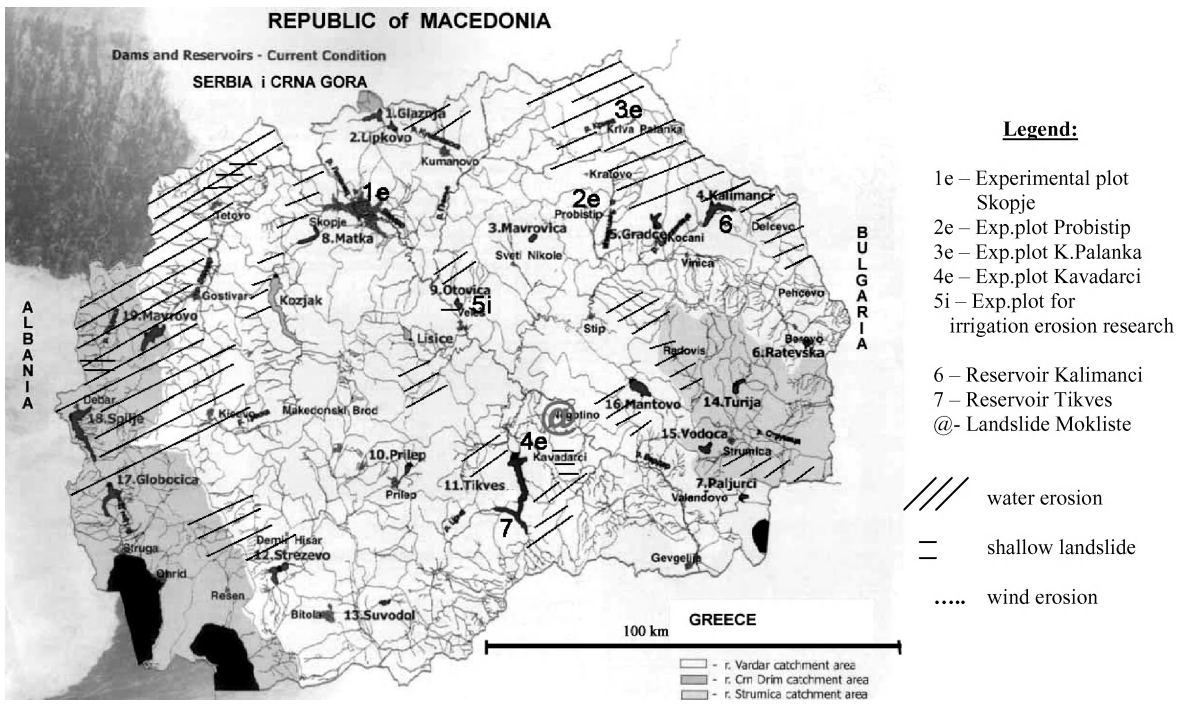

Figure 1.24.1 Erosion processes in the Republic of Macedonia

TABLE 1.24.2 Basic characteristics of the experimental plots

	Skopje (1e)	Probistip (2e)	K. Palanka(3e)	Kavadarci (4e)
Land use	Abandoned arable land			
Slope (%)	11	21	27	16
Bad rock	Palaeogene sediments	Neogene sediments	Gneiss and micaschists	Palaeogene sediments
Soil type	Chromic cambisol	Smolnitza (vertisol)	Chromic cambisol	Colluvial soil
Vegetation cover (%)	90	100	90	95

1.24.3.1 Soil Loss on Experimental Plots

Research has been carried out on experimental plots with an area of 100 m^2 where sediment is collected at the downstream edge and later analysed in a laboratory (Jovanovski and Blinkov, 1992–98). Collectors were perforated and buried in the soil to allow drainage of water while retaining the sediment.

There are four different locations: Skopje (Figure 1.24.1, marked 1e), Probistip (2e), Kriva Palanka (3e) and Kavadarci (4e), with two experimental plots, one of which is natural vegetation (grass, bushes, trees) and the other simulates an arable area. The data (sediment) were gathered during the period 1993–98 several times during the year (Table 1.24.2–1.24.4).

Although there are insufficient data, preliminary relationships between the annual soil loss and the slope were estimated:

$$y = 329.1x - 3884.5 \qquad (r^2 = 0.91)$$

TABLE 1.24.3 Mean annual precipitation [P (mm)] and temperature [t (°C)] during the research period and long period of observation

	Skopje (1e)		Probistip (2e)		K. Palanka (3e)		Kavadarci (4e)	
Period	P (mm)	t (°C)	P (m)	t (°C)	P (mm)	t (°C)	P (mm)	t (°C)
1993–98	404.3	12.9	442.6	12.8	540.2	10.3	497.5	13.7
1951–90	501.7	12.5	471.8	12.8	633.5	10.1	476.7	13.8
Ratio: data 1993–98 (D1)/data 1951–90 (D2)								
D1/D2	0.81	1.03	0.94	1.00	0.85	1.02	1.04	0.99

TABLE 1.24.4 Mean annual soil and nutrient losses on experimental plots

	Mean annual soil and nutrient losses				
Plot	Soil (t ha^{-1})	K_2O (kg ha^{-1})	P_2O_5 (kg ha^{-1})	N (kg ha^{-1})	Humus (kg ha^{-1})
Skopje (1e)	0.040	0.021	0.008	0.172	1.929
Probistip (2e)	3.833	1.598	0.401	1.370	19.226
K. Palanka (3e)	4.773	1.426	0.737	2.704	33.546
Kavadarci (4e)	0.498	0.228	0.109	0.082	2.063

All the results shown are from plots which simulate arable land. Those plots covered with vegetation (grass and bushes) show minimum runoff with no silt in the collectors.

1.24.3.2 Irrigation Erosion

Agriculture is the basis of the Macedonian economy; water deficit is high, especially during the vegetation period, so irrigation is necessary for high quantity yields. There are 106 different irrigation schemes in the country with a capacity to serve 124 000 ha. However, owing to inefficiencies in the systems, not more 80 000 ha are irrigated. Erosion due to inadequate irrigation practices, such as furrow irrigation on sloping land, is less serious than other erosion factors. About 40 000 ha of irrigated land is subject to erosion, with an annual average soil loss of about 308 000 m^3. However, this soil is generally very fertile. About 60% of the irrigation is done through sprinklers and the rest through furrows. Most of the irrigation systems which account for more than 50% of the irrigated land are more than 15 years old and many are seriously damaged. Research was carried out in the period 1975–78 on vineyards near Ve (Table 1.24.5).

1.24.3.3 Landslides and Landfalls

Landslides and falls occur very often. They are natural events, sometimes directly caused by human activities. Usually, landslides occur in settled areas, causing serious damage. The most important landslides occurred in Veles (Ramina), Rostuse and Germo. There are many shallow landslides. During the war period in 2001, a high percentage of the forest in the region of Sar Planina, north-west of Tetovo, was illegaly cut, which caused several landslides in that region.

The biggest landfall on the Balkan Peninsula occured in 1958 in the central part of Macedonia (15 km south of Kavadarci). The eroded material formed a natural dam with a height of more than 35 m. This dammed the River Luda Mara (Crazy Mary) and formed a lake. During the period of high-intensity rainfalls in November 1979, there was a risk of this natural dam breaking and all material being transported, covering with sediment lower parts of Kavadarci city (citizens from vulnerable areas were evacuated). Fortunately, this catastrophe was avoided and now the river is controlled with hydraulic structures. Nearby, in the Tikves region, in 1992, there was a major landslide and several hectares of vineyards were destroyed (this region is marked on Figure 1.24.1 as @).

1.24.3.4 Sediment in Reservoirs

The best way to determine erosion in the catchment area is through direct measurements of the deposited sediment in the reservoir, but for Macedonia this is expensive. Although the Water Act of RM requires annual

TABLE 1.24.5 Results from measurements of irrigation erosion

	Soil	
	Vertisol	Vertisol
Type of irrigation	Furrow	Furrow
Discharge ($l\,s^{-1}$)	0.8219	0.8219
Water velocity ($m\,s^{-1}$)	0.16	0.16
Slope of furrow (%)	3.67	10.10
Soil loss for one flooding ($kg\,m^{-2}$	1.134	3.626
Annual soil loss (3–4 floodings) ($t\,ha^{-1}yr^{-1}$)	39.69	126.91

TABLE 1.24.6 Sediment deposited in two of the most important reservoirs in RM

Reservoir	Catchment area (km^2)	Reservoir storage ($\times 10^6\ m^3$)	Total deposited sediment (m^3)	Mean annual sediment yield ($m^3\ yr^{-1}$)	Mean annual sediment yield ($m^3\ km^{-2}\ yr^{-1}$)
Kalimanci	1135	127	9 413 580	490 629	432
Tikves	5377	475	29 320 019	1 274 783	237

measurement of sediment entry into all reservoirs (now by echo sounder), it cannot be put into practice owing to lack of finance.

For this purpose there were established polygonal nets around the reservoirs and defined cross profiles on the reservoir bed. There is the so-called 'zero situation of the bed' before the reservoir came into use.

There are data from direct measurements for only two of the most important reservoirs: Kalimanci (marked 6 on Figure 1.24.1) and Tikves (marked 7). From 1968, when these measurements started, to 1991 there were nine measurements for each reservoir (Table 1.24.6). In 1991 (year of independence from Yugoslavia), there started a transitional period, with war in the neighbourhood and poverty and low economic standards at home, and this is reflected in the water economy also. In this period, there were only a few measurements spread all over Macedonia, and the results are not used for analyses.

1.24.3.5 Torrent Erosion

A total of 1245 torrents are registered over the whole territory (as a result of the new Water Master Plan) (Table 1.24.7). Torrential flows (flash floods) endanger infrastructural facilities (roads, bridges, etc.) and they cover agricultural land with sterile sediments (stones, gravel, etc.). Small torrents (with catchment areas less than $5\ km^2$) account for 62% of the total number.

Although their catchment area is small, there are torrents with peak flows of more than $30\ m^3\ s^{-1}$ that result in substantial sediment on the flooded areas.

1.24.4 MAJOR ON- AND OFF-SITE PROBLEMS AND COSTS

Owing to the hilly character of RM, soil erosion is spread all over the country. Processes of water erosion are dominant. In the western part of RM, the terrain is rough and steep, so processes of deep erosion are dominant. There are landslides also. In the central part of Macedonia, processes of sheet erosion are dominant. There is wind erosion in this area but the intensity is not as high as water erosion in the other parts. A combination of processes occur in the eastern part of RM. Gully erosion occurs all over the country.

Inadequate ploughing and irrigation have led to different processes of sheet and rill erosion. There are high losses of topsoil, humus and nutrients from the agricultural land located on the slopes.

TABLE 1.24.7 Torrent distribution (degradation categories as in Table 1.24.1)

	Distribution of streams in category of destruction (%)				
Total	I	II	III	IV	V
100	0.48	3.69	39.52	45.70	10.60

Torrential flows are very frequent. There is no city in Macedonia without problems with torrents and their consequences: sedimentation of material in the urban area, damaged streets, bridges, houses and other infrastructure and sometimes with human victims.

Sedimentation in reservoirs is one of the greatest problems. Hence water management enterprises are interested in erosion processes and have invested in anti-erosion measures.

1.24.5 SOIL CONSERVATION AND POLICIES

Measures to control erosion were initiated in the early 1900s, aimed mostly at protecting rivers and reservoirs. Following the passage of the Law on Financing Melioration Systems, these measures were strengthened, and as of 1985, 285 basins were regulated. The water management projections anticipate continuing this work. Given likely budgetary constraints over the coming years, it would be advisable to prioritize these investments.

Measures to control erosion on deforested barren lands have also been under way since 1945, when restrictions were placed on nomadic breeding of goats and sheep in forests. This measure, although unpopular, led to a recovery of degraded forest and shrub land.

There were few Acts directly related to erosion control in the past: the Act for Afforestation of Bareland (1951), Act for Erosion Control on Steep Slopes (1952) and Act for Steep Slope Protection and Torrent Control (1957). Later, these Acts were suspended.

As part of the erosion control programme, an 'Afforestation Fund' was established in 1970. Under these measures, 164 360 ha were afforested, which was 260% more than planned. Since 1990, afforestation has declined 10-fold, mainly because of budget constraints. In an environment of limited available resources, a programme to prioritize areas for afforestation would be useful. There are conflicts in certain areas between afforestation of barren lands and preservation of pastures, even if their quality is poor.

The new Physical Plan of RM (proposal version 2003), predicts afforestation of 80 000 ha of bareland spread over the country. The best way of addressing this issue is through close consultation with farmers and adoption of an integrated approach to watershed management.

Until 1990, anti-erosion measures and activities were on a 'higher level' and institutional support was higher. There were sections for erosion control in all regional water management enterprises. There were parts of the budget aimed at erosion control.

Now, the situation is the opposite. Unfortunately, erosion is one the largest environmental and economic problems in RM, but there is no special Act for erosion control. There are some articles in other regulations (Water Act, Forest Law, Agricultural Land Act, Promotion of the Environment Act), but implementation of these articles is not sufficient for adequate erosion control.

RM ratified the UNCCD convention in 2002. Soil erosion is seen as the most important natural contributor to land degradation and desertification in RM. There is an ongoing project, 'National capacity self assessment for implementation of UNCCD'. A few projects related to UNCCD are planned: national strategy and national and regional action plans.

1.24.6 CONCLUSIONS

Owing to the natural conditions, soil erosion is one of the greatest environmental problems in RM. The vegetation cover is not sufficient to combat erosion processes. Also, some human activities in the last 13 years have contributed to an increase in erosion processes.

Illegal forest cutting, especially in war-affected areas, is a major problem. Forest fires in 2000 destroyed 48 000 ha of forest (4% of the forests in RM). Traditional livestock farming is associated with overgrazing,

particularly with recent increases in the number of goats. Inadequate agricultural practices as a result of farmers' poverty also contribute.

The major problems related to erosion processes are as follows:

1. loss of topsoil (including humus and nutrients), especially from agricultural land located on steep slopes;
2. decreasing soil physical quality;
3. physiological changes and the loss of natural vegetation;
4. disturbance of the water regime, which results in high erosion rates and flash floods;
5. sedimentation in reservoirs.

At the moment 36% of the area is highly eroded and 8% of the area is bare. During the period of socialism, there were numerous anti-erosion activities and soil erosion was reduced, but most activities stopped recently. Erosion is a major problem and one of the priorities for RM is combating this phenomenon.

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1.25

Slovenia

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1.25.1 INTRODUCTION

Very few countries, even considerably larger ones, can boast the landscape diversity found in Slovenia (area 20 273 km^2; population 2 million) since the Alps, the Pannonian Basin, the Dinaric Alps, and the Mediterranean meet and interweave in this small corner of Central Europe (Perko, 1998, 2001b).

1.25.1.1 Geology

Igneous rocks constitute only 3% of Slovenia's surface and metamorphic rocks 4 %. These two types are found primarily in the Pohorje mountains and in the eastern Karavanke mountains. Some 93% of Slovenia's surface is composed of sedimentary rocks. Clay, silt, sand, rubble, gravel and till cover over one-quarter of Slovenia's surface. The greatest amounts of these are found on the flatlands along Slovenia's largest rivers: the Mura, Drava, Krka, Savinja, Sava and Soča. Claystone, siltstone, sandstone, breccia, conglomerate and till cover one-tenth of Slovenia's surface. Marl, which constitutes the majority of the Pannonian low hills, and flysch, which constitutes the Mediterranean low hills, comprise one-tenth of the surface. The same proportion is composed of dolomite, primarily in the Alpine hills. The greatest part of Slovenia's surface, almost one-third, is composed of limestone, particularly the Alpine mountains and the Dinaric plateaux (Verbič, 1998).

1.25.1.2 Relief

Events during the Pliocene and Pleistocene were of major significance for the relief of present-day Slovenia. In the middle of the Pliocene, the surface was largely levelled owing to denudation in a moderately warm and wet temperate climate. Numerous planated areas remain from this period. During the Ice Ages in the Alps and the

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Dinaric Alps, extensive glaciers widened valleys and created enormous quantities of rubble. Water flowing from under the glaciers carried sediments to lower areas. The valleys were deepened by 100–300 m and five to seven levels of terraces, increasingly younger toward the bottoms of the valleys, remain visible on the valley sides (Šifrer, 1998).

In Slovenia one can find six basic genetic types of relief: erosive fluvial, accumulative fluvial, glacial, limestone karst, dolomite karst and coastal. In some areas, these genetic relief types are interwoven (Gabrovec and Hrvatin, 1998).

Almost half of Slovenia is covered by limestone and dolomite karst. In Slovene, the word 'karst' ('*kras*') means 'bare rocky terrain'. The karst surface slowly and imperceptibly dissolves and lowers. The quantity of dissolved limestone carried away by the Ljubljanica River lowers its karst hinterland by 6 mm per 100 yr (Mihevc 1997).

Six relief types are used to categorize surface structure: flatlands, low hills, hills, mountains and low and high plateaux. Flatlands were created by accumulation processes. Today, this process is occurring only on the youngest flood plains. Older gravel terraces are already karstified. The low hills, hills, and mountains are dissected by numerous valleys, but valleys are rare on the karst plateaux where rounded peaks and depressions alternate. Low plateaux extend to 700 m above sea level, whereas the peaks on the high plateaux exceed 1000 m (Gabrovec and Hrvatin, 1998).

Slovenia's average slope is 13 °, the average altitude is 557 m and its highest point is Mount Triglav (2864 m). The snow line in Slovenia is around 2700 m and the tree line lies between 1500 and 1900 m. The average altitude of human settlement is about 500 m below the tree line with the highest farmsteads at 1300 m (Perko, 2001c).

1.25.1.3 Water

Slovenia is rich in water. It has access to the sea, rich reserves of underground water and a dense river network. Owing to the great diversity of relief and rock types, the watercourses are short. While the total length of all watercourses adds up to 26 989 km, as many as two-fifths of them are torrential streams and only 46 are longer than 25 km. Only the Sava, Drava, Kolpa and Savinja rivers are longer than 100 km. Some four-fifths of Slovenia falls in the Black Sea catchment and the rest belongs to the Adriatic catchment (Kolbezen, 1998).

1.25.1.4 Soils

In the Alpine mountains, the prevailing soil is Rendzina; in the Alpine hills, Cambisol; on the Dinaric and the Mediterranean plateaux, Chromic Cambisol; in the Pannonian low hills, Cambisol and Planosol; in the Mediterranean low hills, Cambisol; on the Alpine plains, Rendzina, Cambisol and Luvisol; and on the Pannonian plains, Ranker, Cambisol and Luvisol (Lovrenčak, 1998).

1.25.1.5 Climate

Slovenia has three climate types (Ogrin, 1998): the sub-mediterranean climate, with the average temperature of its coldest month over 0 °C and in its warmest month over 20 °C; the temperate continental climate, with the average temperature in the coldest month between –3 and 0 °C and in the warmest month between 15 and 20 °C; and the montane climate, with the average temperature of the coldest month below –3 °C.

Slovenia has two precipitation regimes: continental and sub-mediterranean. The continental precipitation regime has its peak rainfall in summer. The primary peak of the sub-mediterranean regime is in autumn with a secondary peak between spring and summer. Average intensity over 5 min is from 15 to 20 mm, over 15 min

from 30 to 50 mm and over 24 h it varies from 90 mm in eastern Slovenia to more than 200 mm in western Slovenia (Kajfež-Bogataj, 1996).

1.25.1.6 Vegetation

After the retreat of Pleistocene glaciers, forests covered the entire territory of present-day Slovenia except for the highest and steepest locations. Over many centuries, humans cleared more than a half of the forest. However, forest regrowth has been so intense in recent decades that the proportion of forest cover is approaching two-thirds, thus ranking Slovenia among the most densely forested countries in the world. Most widespread are various beech forests, which comprise almost three-quarters of all forests and cover two-fifths of Slovenia's surface (Perko, 2001c).

1.25.1.7 Landscape Types and Land Use

Based on predominant natural features, Slovenia is divided into Alpine, Pannonian, Dinaric, and Mediterranean landscapes (Perko, 2001a).

The *Alpine landscapes* lie in northern Slovenia, cover two-fifths of its territory and are subdivided into mountain, hill and plain landscapes.

The *Pannonian landscapes* lie in eastern Slovenia and cover one-fifth of its territory. They are composed of densely settled and intensively cultivated areas where forest no longer covers even one-third of the surface. They are divided into the vine-growing low hill landscapes, which are vulnerable to landslides, and agriculturally important plain landscapes along the slow and meandering rivers, which are vulnerable to flooding.

In the south, the *Dinaric landscape* is made up primarily of karst valley systems and plateaux of limestone and dolomite. In contrast to the unfriendly surface is the underground world carved out by water. More than 8000 caves rich with stalactites, stalagmites and other karst cave formations have been discovered so far below the Dinaric and neighboring Mediterranean karst regions.

To the southwest, the Dinaric landscapes meet the *Mediterranean landscape*, which occupies less than one-tenth of Slovenia. It is divided into the more densely populated flysch low hills with their vineyards and orchards and the less densely populated lower karst plateaux (Table 1.25.1). Land use in Slovenia in 2000 is shown in Table 1.25.2.

TABLE 1.25.1 Landscape types (Perko, 1998)

	Area	Average altitude	Average slope	Population		Settlements	
	(%)	(m)	(°)	%	Per km^2	%	Per 100 km^2
Alpine mountains	15.1	1054.5	24.5	4.7	30.1	5.0	9.8
Alpine hills	23.0	582.4	16.9	17.2	72.6	25.2	32.4
Alpine plains	4.0	373.3	4.0	25.1	602.5	6.8	49.8
Pannonian low hills	14.8	288.7	8.8	12.8	84.2	20.8	41.7
Pannonian plains	6.4	196.0	0.8	15.0	228.1	6.4	29.8
Dinaric plateaux	18.8	667.7	13.7	3.3	16.8	13.0	20.5
Dinaric valley systems and plains	9.4	403.3	6.6	12.5	129.4	14.0	44.3
Mediterranean low hills	5.2	305.8	11.1	8.2	151.6	6.4	36.2
Mediterranean plateaux	3.3	426.0	7.7	1.2	35.3	2.3	20.1
Slovenia	100.0	556.8	13.1	100.0	97.0	100.0	29.6

TABLE 1.25.2 Land use by landscape types in 2000 (%) (Hrvatin and Perko, 2003)

	Fields	Vineyards	Orchards	Meadows	Pastures	Forests	Other
Alpine mountains	0.62	0.00	0.34	4.30	5.50	74.85	14.39
Alpine hills	4.07	0.31	1.63	13.09	6.67	68.57	5.65
Alpine plains	29.01	0.03	1.55	15.16	1.77	32.66	19.82
Pannonian low hills	21.77	4.17	2.94	11.09	10.90	41.56	7.57
Pannonian plains	52.72	0.37	0.75	3.57	7.57	21.84	13.18
Dinaric plateaux	1.60	0.36	0.29	2.84	10.15	82.46	2.30
Dinaric valley systems and plains	12.02	0.68	0.63	10.25	15.34	52.49	8.58
Mediterranean low hills	7.96	6.80	3.56	4.20	13.98	55.45	8.05
Mediterranean plateaux	1.60	1.41	0.25	1.00	24.19	68.13	3.41
Slovenia	10.69	1.25	1.28	7.88	9.41	61.49	8.00

1.25.2 HISTORICAL EVIDENCE OF EROSION

In Slovenia, the cultural landscape began to take shape with the first permanent settlements at the end of the Late Stone Age but above all with the clearing of forest for mining needs and new cultivated fields in the Bronze Age and the Iron Age, which resulted in an increase in erosion.

At the beginning of the first millennium AD, Slovenia was conquered by the Romans, who continued clearing forests, establishing cities and building roads.

In the 6th century, the ancestors of modern-day Slovenes largely occupied already cleared areas, and later increased the level of erosion by burning forests to acquire new farmland. Medieval colonization was of major significance, at first directed primarily towards hills and lower mountains with fertile soil and later towards the less advantageous higher hills and mountain areas. Erosion was further increased by the introduction of the plough between the 9th and 11th centuries, but decreased with the introduction of crop rotation. From the second half of the 14th century, the size of the population fell owing to the plague and Turkish raids, and abandoned land increased greatly, occupying almost half the territory of certain regions. The earlier method of the collective colonization of land was replaced by the settling of individuals only in the 17th century (Ciglenečki *et al*., 1998; Mihelič, 1998; Urbanc, 2002).

There are few concrete data regarding soil erosion in the past in Slovenia. We assume that erosion played a somewhat greater role in the past than it does today because the proportion of cultivated areas was substantially larger. According to data from the land cadasters, cultivated fields occupied 18.1% of all surfaces in 1896, but only 12.6% in 1994 (Gabrovec and Kladnik, 1997). In 1896, the proportion of vineyards was 1.0%, of orchards 0.9%, of meadows 15.9%, of pastures 17.0% and of forest only 41.6%, and the proportion of built-up, water and other surface areas was 5.0%.

According to Vrišer (1953), areas composed of young and poorly cemented marl and sand sediments in the Pannonian hills and flysch in the Mediterranean hills were most affected by soil erosion. He is convinced that in past centuries, many farms established at the height of the medieval colonization in the 14th and 15th centuries were abandoned owing to soil erosion.

Owing to great sheet and tillage erosion, the farmers in hilly regions had to carry the soil that was washed away back to the fields in baskets.

In Slovenia, summer storms with heavy precipitation are the most frequent cause of soil erosion. In Haloze, a region of irregular Pannonian low hills, heavy erosion on sloping vineyards washed the already meagre soil down to the valleys. To protect it as best they could, farmers began to grow various kinds of cover plants between the vines, and elsewhere the soil was covered with straw.

Figure 1.25.1 Cultivation terraces (photo: Stane Klemenc). The construction of terraces for cultivation has been one of the most frequent measures to prevent water erosion on hillslopes. (Reproduced by permission of AMGI ZRC SAZU)

The damaging effects of erosion were substantially reduced with the introduction of terraced vineyards (Bračič, 1967). The use of terraces for cultivation has long been known on the Slovene coast and its immediate hinterland. Although there is no direct evidence indicating the construction of terraces for cultivation in the Roman period, this is highly probable as otherwise it is difficult to imagine growing grapes and olives, the main agrarian activities in the Roman civilization. In later historical periods, the construction of terraces spread here with the increasing needs of the growing population. All the work of building terraces was done by hand, and there was enough labour available in this region until the end of the 19th century (Titl, 1965) (Figure 1.25.1).

The construction of terraces for cultivation brought many positive consequences. Water flowing down the slopes during rain settles on the terraces and no longer carries away fertile soil. At the same time, a great deal of moisture accumulates in the soil, which, during the growing period, often eases the impact of droughts. The recognition that on a sloping surface mechanization could only be of use on sufficiently large cultivated terraces further encouraged the construction of terraces. The majority of terraces along Slovenia's coast lie on slopes with inclinations of 20–40°. In these conditions, many are narrow and in places very short. In spite of this, the greater part of the farm produce was grown on these terraces. Today, numerous terraces are abandoned, for various reasons. For example, the proportion of the farming population decreased greatly, machine cultivation of small parcels is difficult and often unprofitable and the annual maintenance of these terraces is demanding and time consuming.

A unique type of soil erosion is found in karst landscapes (Gams, 1974). Clearing of forests to form meadows or cultivated fields changed the appearance of the karst landscape. Cultivation, however, greatly accelerated the washing away of the soil. Few farmers were aware of the intensive denudation, although most of them knew that rock 'grows' on karst soil. Water washes the upper layer, particularly the smallest particles, into the newly formed cracks while sheet erosion during downpours occurs on slopes, particularly in vineyards, which is why abandoned vineyards are barren in places. In areas of the Mediterranean karst where the bora strikes, the wind also carries away the soil if there is no snow or if the ground is not covered by grass (Figure 1.25.2).

The speed of soil erosion in the karst region is a controversial question. Hrovat (1953), who described the erosion in the fields and vineyards of south-eastern Slovenia in detail, established that on average the soil is

Figure 1.25.2 Former karst meadow (photo: Mimi Urbanc). In a former karst meadow, the height of the limestone outcrops above the ground equals the extent of the soil erosion. (Reproduced by permission of AMGI ZRC SAZU)

lowered by 1 cm yr^{-1}. On the basis of his results, Hrovat believed that Slovenia's karst was on the path that the Mediterranean karst had already walked, towards becoming a rock desert. However, Hrovat's evaluation of the intensity of the soil erosion is greatly exaggerated and possibly only applies to the erosion in the most vulnerable areas.

Gams (1974) developed a methodology for the indirect determination of erosion in karst areas. He realized that numerous rocks projecting on the surface still show signs of being cut off at the top. If the rocks were cut off in a former meadow, the height of the rocks above the ground equals the extent of the soil erosion. If the rocks were cut off in a formerly cultivated field, we have to add as many centimetres as the rock was cut off deep under the ground. A second method offers another alternative. Solid limestone has a smooth surface at the contact point with soil. The surface may be uneven, the stone may have various shapes, but the rock surface is not rough. Rocks that have been projecting from time immemorial, however, are full of cracks and fissures and their surface is rough. The trained eye can see whether the surface relief on the rock originated in the soil or above it, even though many decades have passed since the lowering of the soil.

Areas in different parts of Slovenia studied using both methods showed great differences relative to erosion. On flatter meadowland, the rocks usually project only 20–30 cm from the ground, and the extent of the erosion is thus the same. In vineyards, erosion is normally much greater. Gams presumes that soil erosion after the clearing of forests and the first ploughing was rapid and later gradually slowed.

Even though the denuding of the soil is not as rapid as Hrovat claimed, erosion remains an underestimated factor. Fortunately, land in Slovenia's karst region is increasingly less cultivated and cultivation is increasingly concentrated at the bottom of karst poljes, dry dolines and uvalas where the soil is thicker.

1.25.3 CURRENT EROSION PROCESSES

Areas with steep inclinations and areas of less resistant rock such as shale, siltstone, sandstone and marl are especially vulnerable to erosion. In the Pannonian low hills, erosion appears to a greater extent in particular on Tertiary marl and sandstone where landslides and slumps are frequent. In the Mediterranean low hills, steep

Figure 1.25.3 Landslide area (photo: Maja Topole). The Pannonian low hills are composed of low-resistance rocks, primarily marl, sand and clay, and are therefore vulnerable to landslides. (Reproduced by permission of AMGI ZRC SAZU)

flysch slopes are most vulnerable to erosion. Torrents threaten especially the Alpine mountain chains and hills and the Pannonian low hills.

There is little information about the intensity of erosion on farmland in Slovenia. This is most probably due to the lack of awareness of this phenomenon and the fragmentation of the land: in 1991 the average Slovene farm comprised only 5.9 ha of land (Kladnik 1998). Somewhat more information has been published on fluvial erosion. In Slovenia, only one measuring station, located in the village of Smast near Kobarid in the Soča Valley, was in operation for a long period in the second half of the 20th century (until 1989). Indirect data were also provided by measurements of the carrying power of water courses and of the amount of suspended material in the reservoirs of hydroelectric plants on Slovenia's major rivers, the Soča, the Sava, and the Drava.

Landslides occur on about 30% of the surface area of Slovenia (*National Programme*, 2002), snow avalanches threaten 0.5% (9820 ha) (Pavšek, 2002) and glacial erosion today only occurs on less than 3 ha (Figure 1.25.3).

1.25.3.1 Wind Erosion

Wind erosion is spatially limited to south-western Slovenia. The strongest wind is the bora, which blows from the southern edges of the high Dinaric plateaux across the Mediterranean low hills and plains. In the period between 1975 and 1985, its maximum reported velocity was 170 km h^{-1}, its average annual velocity 19.5 km h^{-1} and the average intensity of gusts 94.5 km h^{-1} (Mihevc 1997). The bora is strongest in the Vipava Valley where extensive land improvement was carried out in the 1980s. Belts of bush and hedges between pieces of land were removed, which further accelerated wind erosion. The erosion was most pronounced in the year following extensive drainage projects when many ploughed pieces of land were left bare over the winter (*Poročilo*, 1996).

The bora is also a frequent occurrence in the coastal hills. In February 1954, its erosive effects were observed in the hinterland of Koper. In some places, a bora with a maximum speed of 23.7 m s^{-1} removed up to 10 cm of soil, in some places right down to the roots of the vines. Because of strong wind erosion, mainly land in protected sites was cultivated in the past, while pasture and forest dominated in windy areas (Malovrh, 1955).

Figure 1.25.4 Landslide (photo: Milan Orožen Adamič). In Haloze (eastern Slovenia), more than 5000 slumps and small landslides were triggered in an area of 106 km^2 during a major storm between 3 and 4 June 1989. (Reproduced by permission of AMGI ZRC SAZU)

1.25.3.2 Landslides and Slumps

In Slovenia, landslides and slumps are also triggered on farmland. Between 6000 and 8000 landslides are presumed to occur, but only about 3400 of these are recorded, of which around 1000 present a threat to transportation routes. Landslides ordinarily not only encompass soil but also reach deeper and therefore occur not only on farmland but also in forests.

Landslides can potentially occur in one-third of the territory of Slovenia. Larger areas are often affected. In Haloze in eastern Slovenia, more than 5000 slumps and small landslides were triggered in an area of 106 km^2, which is ca 47 km^{-2}, during a major storm between 3 and 4 June 1989 (Natek, 1990) (Figure 1.25.4).

1.25.3.3 Water Erosion

On cultivated farmland, erosion gradually decreased in the past with the abandonment of farming on steep surfaces and the abandonment of cultivation due to social and economic factors. In the last decade, it has been observed that the erodibility of the soil is also influenced by climate change, primarily by longer periods of drought that reduce the infiltration ability of the soil and by heavy rains that increase runoff. In some places, intensive farming has caused the compaction of the ground, which decreases the infiltration ability of the soil and increases surface runoff and erosion (*Poročilo*, 2002).

Degradation of the soil also occurs owing to illegal construction, clearcutting, unmaintained forest roads and unprofessionally prepared ground work for pastures or vineyards (*Poročilo*, 1996).

In the Slovene literature, various data appear regarding the total area of erosion. In most cases, authors state values between 42 and 44% of Slovenia or between 8800 and 9000 km^2 (Zemljič, 1972; Kolbezen, 1979; Horvat, 2002).

Torrential erosion presents a threat to almost one-fifth or 4000 km^2 of Slovenia. Dolomite areas of the Alpine mountains and hilly and low-hill areas on less resistant non-carbonate rock are vulnerable. In the territory of Slovenia, there are about 370 erosion focal points and 700 major torrents. There are more than

1700 km of torrential streams and 4000 km of their tributaries have lengths over 0.5 km, which gives a density from 1.6 to 3.5 km km^{-2}. This is almost three times the density on flat areas, with about 0.5 km km^{-2} (Rainer and Pintar, 1972; Zemljič, 1972).

Annually, an average of 2.5×10^6 m^3 or 10 t ha^{-1} of material is released from these areas. Erosion totals 45 t ha^{-1} yr^{-1} in the mountainous region of the western Karavanke Mountains and 50 t ha^{-1} yr^{-1} on the Soča River side of the Julian Alps (Zemljič, 1972).

Annually, about 15 t ha^{-1} of material is deposited in the rivers and streams of the Soča river basin, around 6.3 t ha^{-1} in the Sava river basin, around 5.7 t ha^{-1} in the Drava river basin and around 2.6 t ha^{-1} in the Kolpa river basin. In the coastal hills, about 6.4 t ha^{-1} is deposited annually in the rivers and streams (Zemljič *et al.*, 1970).

In the anthropogenically degraded landscape of the Meža Valley in the eastern Karavanke Mountains, the vegetation was completely destroyed following lead pollution. During the operation of the mine's separation plant, erosion washed away on average more than 110 t ha^{-1} of material annually from a 0.5 km^2 area of dolomite.

In Slovenia, regulatatory measures are needed on over half (2370 km^2) of the torrential surface areas (Horvat and Zemljič 1998). Torrents often present a threat to settlements and infrastructure, particularly roads, since on average a torrent crosses a road every 834 m (Horvat, 2002). Torrents contribute a significant amount of sediment to reservoirs. Assessments of the loss of material for all of Slovenia most frequently range between 5×10^6 and 6×10^6 m^3 annually (Rainer and Pintar, 1972; Kolbezen, 1979). The majority of authors give figures between 5 200 000 and 5 300 000 m^3. The figures were calculated using the Gavrilović method, which was adapted to Slovene conditions (Zemljič, 1972; Rainer and Zemljič, 1975; Horvat, 1987, 2002; Horvat and Zemljič, 1991). Average removal of material is about 5 t ha^{-1} yr^{-1}. Some authors suggest even lower figures for annual removal of material, for example Lazarević (1981) with 3 960 200 m^3 or about 4 t ha^{-1} yr^{-1}.

Approximately three-fifths of the removed material is deposited on slopes, talus and fans, and also in erosion and torrential gullies. The remaining material reaches streams and rivers, but about one-quarter stops at the transitions from torrents into rivers and in their upper courses. Owing to the deposition of material, the bottoms of the riverbeds constantly rise, gravel beds spread at the expense of other land and the danger of flooding increases (Zemljič, 1972).

1.25.4 EROSION ON FARMLAND

Long-term measurements of soil erosion on farmland were carried out on only one test field; elsewhere, only shorter observations were made. Measurements were made in the village of Smast in the Soča Valley, Latkova vas in the Savinja Valley, Straža in the Krka Valley, in the Dragonja Valley near the coast and in the Mirna Valley. The measurements (Tables 1.25.3–1.25.6) indicate that erosion on bare soil on farmland totals about 20 t ha^{-1} yr^{-1} and on overgrown fields around 4 t ha^{-1} yr^{-1}. On surfaces with the same inclination near Smast, the measured erosion was only 6.3 kg ha^{-1} yr^{-1} in a mixed forest, 39 kg ha^{-1} yr^{-1} on a meadow, 3.5 t ha^{-1} yr^{-1} on a potato field and as much as 22.4 t ha^{-1} yr^{-1} on a ploughed field (Horvat and Zemljič 1998).

Determining erosion using the GLEAMS 2.1 model in Latkova vas indicates that the erodibility on a hop field with an inclination of 0.4% is about the same as the erodibility on a potato field with a surface inclination of 55.7% near Smast. In 1997, the erosion on a hop field near Latkova vas reached almost 5 t ha^{-1} yr^{-1}. Growing hops, which is characteristic of the Celje Basin, presents a relatively large threat to the soil (Zupanc *et al*. 2000).

Ravbar (1975) performed two measurements of erosion on karst clay near Straža. He observed the loss of soil when 36 mm of precipitation fell and during a downpour when 107 mm fell. During the first event, 1160 g

TABLE 1.25.3 Soil erosion at the Smast experimental station in the Soča Valley in western Slovenia between 1 October 1972 and 27 July 1977, on brown carbonate-based soils and plots of 50 m^2, where average annual precipitation is 2699 mm (*Klimatografija Slovenije*, 1995) (Horvat and Zemljič, 1998)

	Slope(%)	Runoff	Runoff ($l\ m^{-2}$)	Runoff ratio	Soil loss(g)	Soil loss ($g\ l^{-1}$)	Soil loss ($t\ ha^{-1}$)	Soil loss ratio
Fallow	55.7	10383	239.79	61	97022	9.344	22.40	3556
Potato field	55.7	4609	106.44	27	15049	3.265	3.47	551
Grassland	55.7	1033	23.85	6	169	0.164	0.0390	6
Mixed forest	54.7	173	3.93	1	28	0.161	0.0063	1

TABLE 1.25.4 Annual losses in Latkova vas in eastern Slovenia between 1997 and 1998: data acquired on a hop field with sandy-clay soil on the basis of the GLEAMS 2.1 model (Zupanc *et al.*, 2000)

Year	Slope(%)	Annual precipitation(mm)	*K* (erodibility coefficient)	Loss ($t\ ha^{-1}$)
1997	0.4	1235	0.5	4.88
1998	0.4	1170	0.5	2.39

TABLE 1.25.5 Soil erosion in the Mirna Valley, south-eastern Slovenia, calculated using the USLE and a digital model of altitude (Topole, 1998)

Relief units	Proportion of relief units relative to entire watershed area (%)	0.5–4.9 t ha^{-1}	5.0–14.9 t ha^{-1}	15.0–34.9 t ha^{-1}	35.0–74.9 t ha^{-1}	75.0–149.9 t ha^{-1}	150.0–299.9 t ha^{-1}	300.0–599.9 t ha^{-1}	>600.0 t ha^{-1}
Low hills	43.76	5.96	19.37	28.76	26.45	12.96	5.07	1.27	0.16
Hills	40.06	4.65	15.79	30.95	30.27	12.58	4.67	1.01	0.08
Basin	13.28	20.86	36.93	24.25	12.75	3.93	1.05	0.23	0.00
Plateau	2.91	12.47	37.19	30.30	13.40	5.59	0.82	0.23	0.00
Total	100.00	7.60	20.79	29.08	25.78	11.40	4.25	1.00	0.10

TABLE 1.25.6 Comparison of the Gavrilović and RUSLE methods for various land uses in the Dragonja river basin in southwestern Slovenia on eutric brown soil on flysch (Petkovšek, 2002)

	Annual soil loss (t)		Annual soil loss ($t\ ha^{-1}$)			
	RUSLE	Gavrilović	RUSLE	Gavrilović	Average slope (%)	Area (ha)
Fields	6,322	3,205	21.6	10.94	19	293.1
Orchards	309	69	20.88	4.77	33	14.4
Vineyards	2,208	952	51.31	22.12	21	43.0
Meadows	627	610	4.80	4.67	28	130.7
Pastures	764	427	3.39	1.89	36	225.5
Forests	298	54	2.55	0.46	52	116.9
Other	80	72	4.92	4.43	30	16.3

of material was eroded, and during the second 290 g. From the data on the erosion and the fact that the average annual precipitation in this area is 1138 mm (*Klimatografija Slovenije*, 1995), we calculated that soil erosion here on average is 22 t ha^{-1} yr^{-1}.

Soil erosion was also modelled in the Dragonja watershed where the average annual precipitation is 1017 mm (*Klimatografija Slovenije*, 1995, 47). According to land use, using the Gavrilović method, the calculated erosion in vineyards was 22 t ha^{-1} yr^{-1} and on fields 11 t ha^{-1} yr^{-1}, and according to the RUSLE method, in vineyards 51 t ha^{-1} yr^{-1} and on fields 22 t ha^{-1} yr^{-1} (Petkovšek, 2002).

Calculations for the Mirna Valley made on the basis of the USLE indicate that erosion on average is as much as 52 t ha^{-1} yr^{-1}. On more than half of the area studied, the calculated erosion was lower than 35 t ha^{-1} yr^{-1}, and on less than one-fifth of the area, greater than 75 t ha^{-1} yr^{-1}. Owing to the less resistant rock and in spite of lower altitude, the low hills are more vulnerable to erosion than the hills (Topole, 1998).

Using the Gavrilović method, Mikoš and Zupanc (2000) established that the surface on cultivated field areas in Slovenia is lowering on average from 5 to 10 mm yr^{-1} and that the erosion is between 8 and 16 t ha^{-1}.

The measurements offer an insight into the intensity of the erosion of farmland, which is approximately three times lower than on areas affected by torrents. Erosion reaches higher values particularly on cultivated surfaces with steeper slopes and where the soil is bare. On the basis of the data collected, we can state that the type of soil, the inclination of the surface and the level of precipitation especially influence soil erosion. The type of vegetation cover and the method of cultivating the soil are also significant factors.

No measurements have been made in Slovenia on the basis of which the ratios between gully, rill, and sheet erosion on cultivated land can be established.

For Slovenia, erosion on farmland is less important from the viewpoint of the national economy than torrential erosion since the latter more often affects costly road infrastructure than farmland, where more damage is caused by drought and floods than by erosion.

1.25.5 SOIL CONSERVATION

Owing to the reduction in farmland, the quantity of eroded material has correspondingly decreased. Since the 1960s, industrialization and urbanization have encroached on arable land (Gabrovec and Kladnik, 1997). Farmland on plains with fertile land has been reduced by the construction of new expressways.

In the 1970s, the use of almost 1000 ha of farmland in Slovenia changed annually, and in the 1980s about 500 ha annually. After 1990, changes in the use of land again increased. Between 1993 and 1997, built-up and road areas increased by 4078 ha and farmland decreased by 81092 ha (*Poročilo*, 2002).

Slovenia's 1996 *Law on Agricultural Land* defines measures for the protection of farmland. However, it only considers agro-ameliorations, hydro-ameliorations and land consolidation, which is why no funds are provided for protection from erosion in the national budget. According to this law, landowners must adapt farming production to the ecological and soil conditions and use suitable methods to prevent erosion.

The law does not prescribe the kind of anti-erosion methods that farmers should use on farmland. In reality, farmers use very diverse methods, which is connected with fragmentation of land and the ownership situation. A great deal of land is owned by part-time farmers who only cultivate their land in the afternoons after their regular jobs.

Among the old methods for preventing erosion used by farmers and part-time farmers in Slovenia are primarily the selection of suitable field crops, crop rotation and the planting of cover plants between rows. Water erosion is more pronounced on maize with larger distances between individual plants and less on the fields with densely sown grass or clover. The erodibility of the soil is only rarely lessened deliberately by reducing the cultivation of the soil, mulching and sowing immediately after a harvest. Particularly in the coastal hills, belts between individual types of field crops are grassed over. In eastern Slovenia, drainage

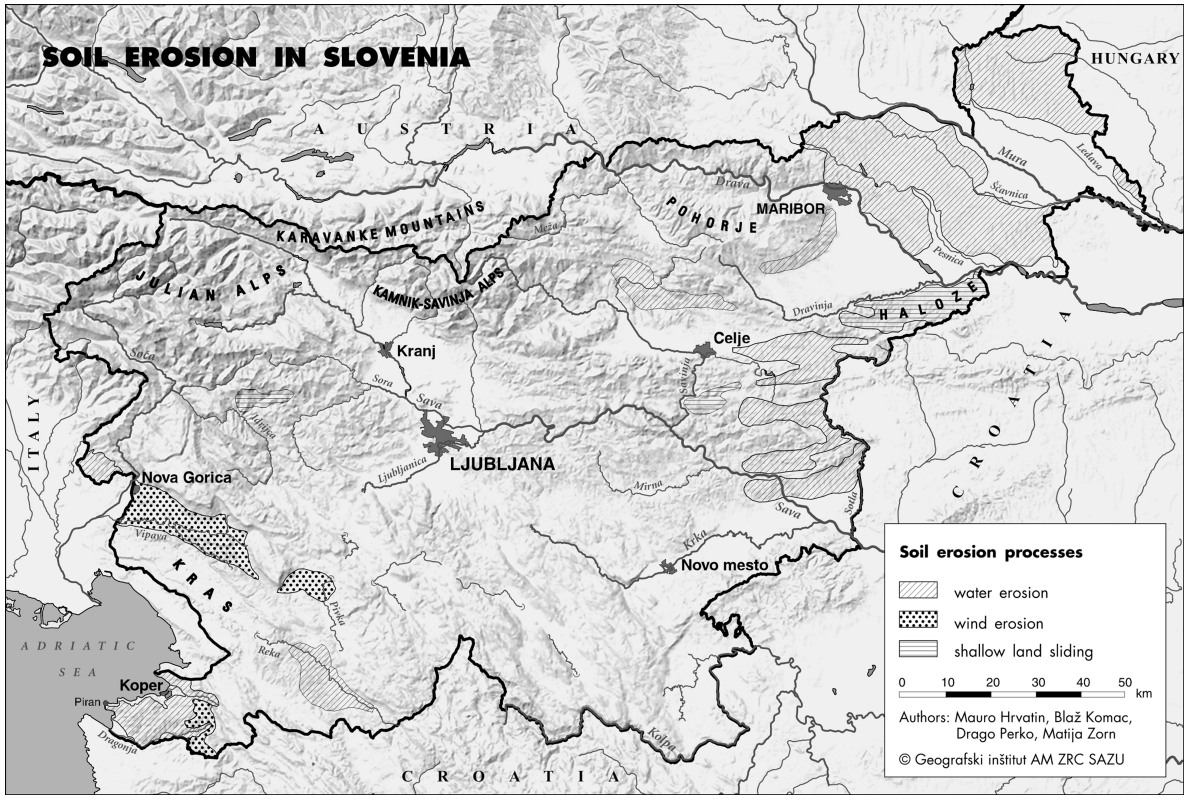

Figure 1.25.5 Soil erosion in Slovenia. (Reproduced by permission of AMGI ZRC SAZU)

ditches are dug on farmland to redirect the flow of the water. Owing to the fragmentation of land, boundary hedges and intervening belts of grass or bushes are frequent, which reduces the speed of the water flowing off the surface. In fruit growing and vineyards, terracing and contour ploughing are often used to intercept the runoff, hold the water and encourage deposition of eroded material. Erodibility is also reduced by the proper use of suitable machinery (Zupanc and Mikoš, 2000; Zupanc *et al.*, 2000) (Figure 1.25.5).

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1.26

Spain

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Spain is one of the countries most severely affected by soil erosion in the European Mediterranean region owing to extreme spatial and temporal variations in its physical environment, with frequent periods of drought and torrential rainfall. The presence of soils that are highly erodible because of their poor organic matter content and weak structure, and uneven relief with steep slopes, explains why a large part of Spain has a high erosion risk potential. Furthermore, a long history of anthropogenic disturbances related to temporary increases in population and its pressure on marginal lands are key factors in accelerating soil erosion processes and increasing soil loss.

1.26.1 THE PHYSICAL ENVIRONMENT

Spain occupies the largest part, about 504 000 km^2, of the Iberian peninsula, in south-western Europe (Figure 1.26.1), and similarly to other Mediterranean countries, is characterized by a particular combination of climatic, lithological, topographic and historical land-use factors that favour widespread soil erosion. The Balearic Islands in the Mediterranean have similar geographic conditions and are also treated in this chapter.

1.26.1.1 Climate

Spain can be divided into two large climatic sections (Figure 1.26.2): the humid Spain of the north, down to the central plains, called the Meseta, with a warm oceanic climate, continuously mild tempertures, and rain throughout the year, and the dry Spain, with warm winters and hot, dry summers, although Rivas Martinez (1987) found a large variety of subclimates. About 80% of the Spanish climate is Mediterranean (between

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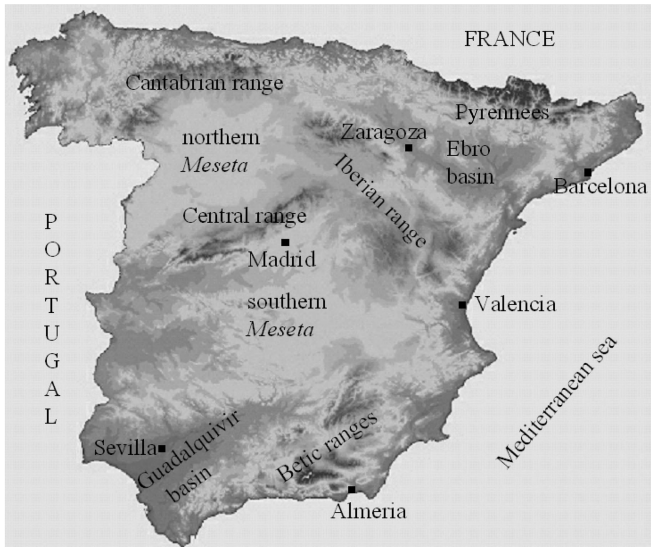

Figure 1.26.1 General map of Spain, with main cities, mountain ranges and large basins. Hypsometric colours: grey, 0–500 m; light grey, 500–1500 m; dark grey, above 1500 m

semi-arid and sub-humid) with annual precipitation ranging from 300 to 600 mm, often in the form of local, conventional storms concentrated in autumn and spring, with frequent summer droughts. The spatial distribution of maximum rainfall intensities in 1 h is shown in Figure 1.26.3. Most of the country has rainfall intensities above 30 mm h^{-1} lasting 1 h, and in many spots along the Mediterranean coast such storms even exceed 70 mm h^{-1}. Up to 200 mm h^{-1} lasting for 10 min has been recorded in Valencia and a few other spots with a return period of 50 years (Elias Castillo and Ruiz Beltrán, 1979). Mediterranean rainfall can reach extremely high absolute values and two remarkable cases are given as examples: (a) in Gandía, in the Region of Valencia, over 1000 mm in 36 h were recorded in November 1987 and over 400 mm fell in less than 6 h (López-Bermúdez, 1992); (b) on 7 August 1996, an extreme rainfall event produced a devastating debris flow in a mountain catchment in the Central Pyrenees with a maximum rainfall intensity estimated at 515 mm h^{-1} for 8 min in a 2-km^2 subcatchment (White *et al*., 1997).

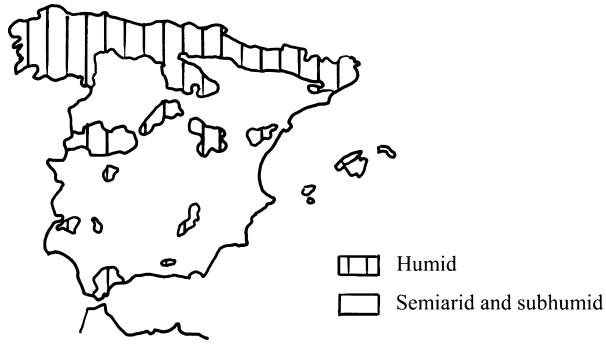

Figure 1.26.2 Climatic zones of Spain (except the Canary Islands), according the UNESCO aridity index (P/ETP): humid (>0.75), subhumid + semiarid (<0.75)

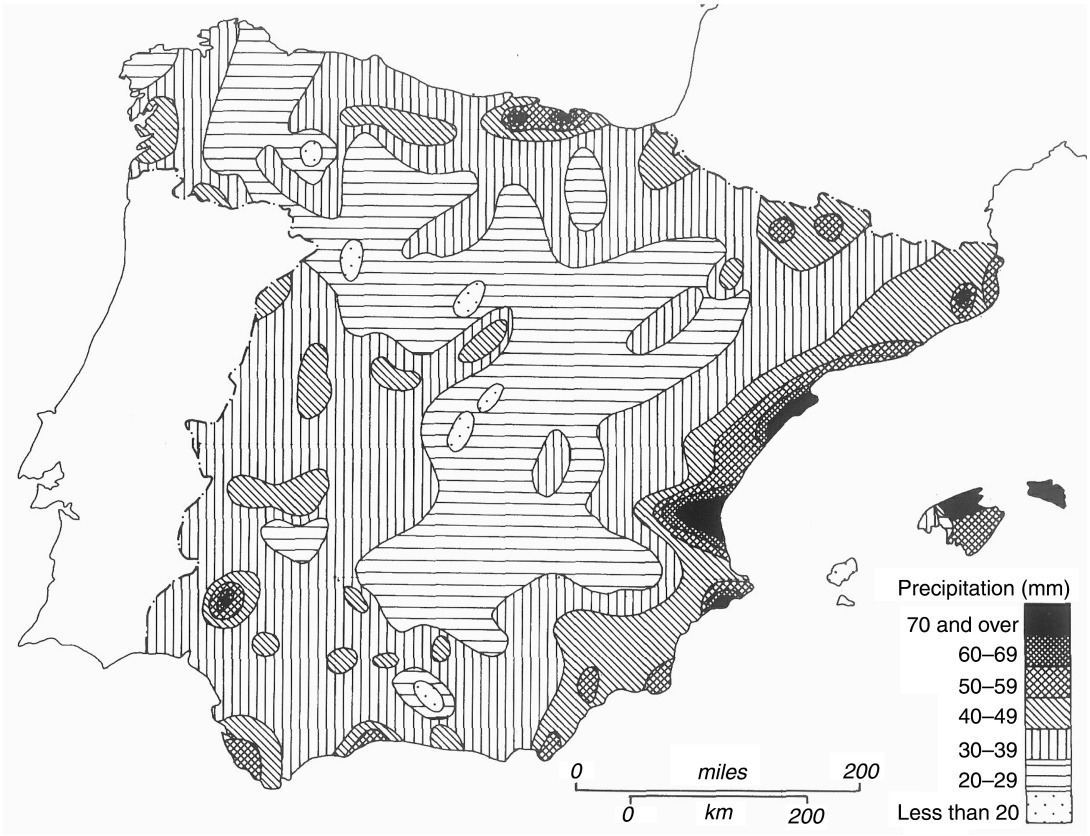

Figure 1.26.3 Distribution of maximum rainfall intensities in 1 h, according to Elias Castillo and Ruiz Beltrán (1978), in Grove and Rackham (2001). (Reproduced from Grove AT and Rackham O, The *Nature of Mediterranean Europe. An Ecological History*, Copyright 2001, Yale University Press)

As in the rest of the Mediterranean, interannual variability in precipitation and temperature are and have been important, even during the cold fluctuation of the Modern Age (Creus *et al*., 1990, cited by Puigdefábregas, 1995). However, droughts are not so widespread as in sub-Saharan regions, and are discontinuous both spatially and temporally. Under such conditions, the amount of natural vegetation is sparse and contributes little organic matter to soils, and as a consequence provides little protection from rain showers in the form of plant cover and/or well-aggregated soils.

1.26.1.2 Geology, Lithology, Physiography

Spain is a very mountainous country, with an average altitude of around 660 m (the second highest in Europe after Switzerland). This great height is due to extensive high plains, the largest in Europe, located mainly in the centre of the country, which correspond to the structural surfaces of tertiary interior basins and erosion surfaces, with occasional residual relief. All this together constitutes the Meseta, which is surrounded by *cordilleras*, or mountain ranges (Cantabrian, Iberian and Betic ranges). Most of these mountain ranges are young and often tectonically active, and most of them very close to coastal areas (Figure 1.26.1), where population is concentrated. During

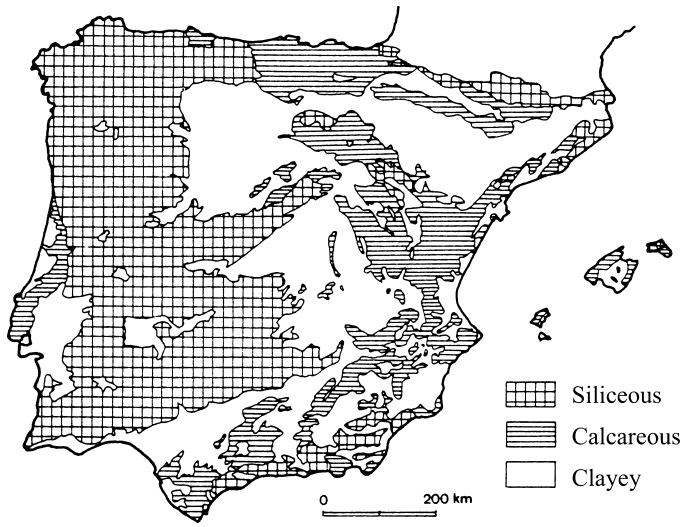

Figure 1.26.4 Main lithological units of the Iberian peninsula, according to Solé-Sabaris *et al.* (1952), in Gutiérrez Elorza (1994). (Reproduced from Gutiérrez Elorza M, *Geomorfología de* España, 1994, Copyright Editorial Rueda)

recent millenia, some coastal ranges have experienced uplift of from 50 to 1000 mm kyr^{-1}. Most coastal regions have been shaped since the Pleistocene or even older ages: neotectonics is one of the main agents of change in the present landscape, along with climatic fluctuation and one of its consequences, sea level oscillation, is responsible for the erosion–stabilization stages of most Mediterranean landscapes during the Quaternary. All this complex geology and topography determine a large variety of fluvial regimes (Masachs, 1948) and in many of them hillslopes and drainage networks are far from equilibrium and give rise to high rates of erosion and sedimentation.

Maps of the dominant lithologies in the Iberian Peninsula (Figure 1.26.4) (Gutiérrez Elorza, 1994, based on Solé Sabarís, 1952; Riba, 1969) coincide rather well with the map of the main morphostructural units (Gutiérrez Elorza, 1994; Gutiérrez and Casares-Porcel, 1994) (Figure 1.26.5), in which it can be observed how the western and central sectors of the country that make up the Hercynian base of the Meseta are mainly formed by hard, resistant materials (plutonic rock, gneiss, quartzite, schist). On the other hand, the peripheral Alpine mountain ranges present, in addition to their young relief, a softer lithology, with sedimentary rocks such as sandstones. Moreover, most of the large tertiary basins are made up of soft materials, some of them, such as marls, mudstones and shales, having a strong tendency to gullying, in both humid (Regües *et al*., 1995) and dry conditions (Gutiérrez Elorza *et al*., 1995; Gallart *et al*., 2002).

1.26.1.3 Soils and Land Use

The climatic factor determines soil and land-use behaviour through the water balance, which can be estimated from the difference between precipitation and potential evapotranspiration. Soils in the south-east have a permanent water deficit and those of the Pyrenees show little deficit or even excess. In most Mediterranean regions, complete leaching of the soil profile never occurs, only partial (Roquero, 1992). The most common soil water regime in the Mediterranean sector of Spain is *xeric* (Soil Survey Staff, 1999), but also the *aridic* regime (Soil Survey Staff, 1999) is found in the south-east, and consequently Xerochrepts and Xerorthents are the most common soils and also those most vulnerable to erosion (Roquero, 1992). Next are Torriorthents (local but even more susceptible to erosion). With regard to Aridisols, characterized by either the aridity of

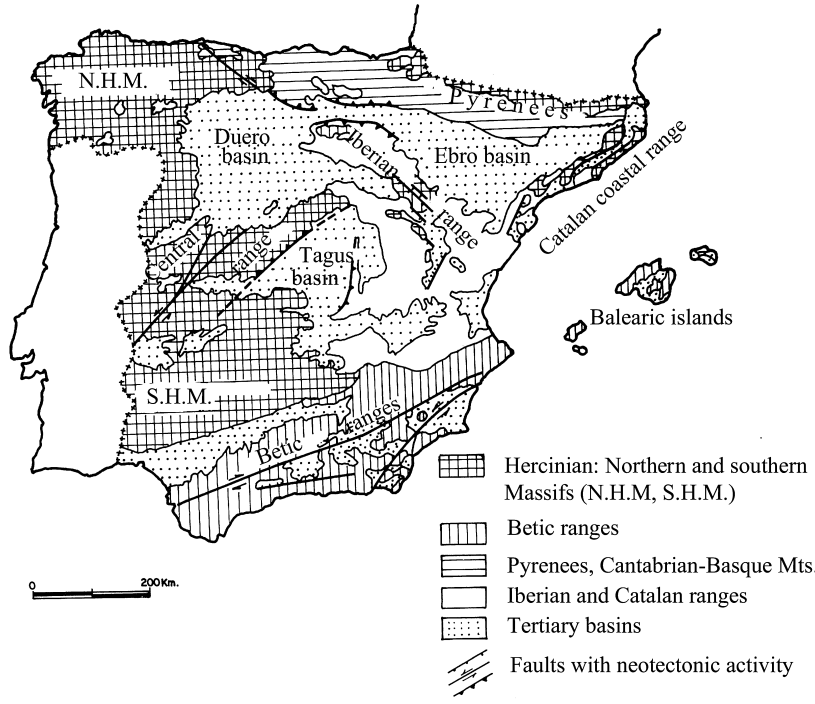

Figure 1.26.5 Morphostructural units of the Iberian peninsula, according to Gutiérrez Elorza (1994). (Reproduced from Gutiérrez Elorza M, *Geomorfología de España,* 1994, Copyright Editorial Rueda)

climate or high salinity, they are fairly frequent in the south-east and in the Ebro valley, and represented by Calciorthids, Camborthids and Paleorthids, all vulnerable to erosion. Also Argids, relict soils from Central Spain, although not occupying a large area, suffer from severe erosion. The fourth soil order is Alfisols, represented by Xeralfs, soils with an argillic horizon, fairly abundant in fluvial terraces and *rañas* in the centre or the country, and less susceptible to erosion. Within Vertisols, only Xererts are found, and within this sub-order, Chromoxererts have erosion problems. Finally, from the order of Ultisols, Xerults, fairly dispersed in central Spain, show erosion problems at the edges of platforms (Roquero, 1992).

Within the humid part of the country, most soils have *ustic* and *udic* moisture regimes (Soil Survey Staff, 1999), with a surplus water balance through the year, and consequently the complete leaching of soil profiles determines soil formation, evolution, properties and behaviour. Under these conditions, in northern (Asturias, Cantabria and part of Euskadi) and north-western Spain (Galicia), acid and podzolic soils occur.

Finally, the least represented order is that of Histosols, soils with a large accumulation of organic matter poorly evolved, present only in high mountain areas (Pyrenees, Central, Iberian and Cantabric ranges).

Except for high mountain areas, most soils in Spain have been cultivated for some time and the long sequence of agricultural clearing and abandonment has been common at least in recent millennia and accelerated with the increase of population in the first half of the 20th century. However, the trend during the second half of the 20th century has been agricultural abandonment due to mechanization.

Recent decades have witnessed an accelerated land-use change: in the period 1975–96 there has been a 10% decrease in the total cultivated surface (16% of rainfed lands, although irrigated land has increased by 26%). Also pasture has decreased (16% in rainfed land), in addition to forest (23%) (MAPA, 1999) (Table 1.26.1).

TABLE 1.26.1 Total surfaces (ha) of different land uses from 1975 to 1996 in Spain

	Cultivated land		Natural pastures			
Year	Dryland	Irrigated	Dryland	Irrigated	Steppe	Timber forest
1975	18 216 800	2 616 800	1 305 200	201 200	5 719 400	9 395 900
1985	17 409 000	3 006 400	1 246 600	210 500	5 270 700	7 252 100
1996	15 342 700	3 309 600	1 095 200	292 900	5 578 600	7 240 800

As indicated above, climatic characteristics determine to a large extent the dryness of the majority of Spanish soils, with low levels of organic matter, a weak structure and consequently a fairly high level of erodibility. Many Spanish soils show the main indicators for soil erosion (Figure 1.26.6): shallowness, low organic carbon content, low water holding capacity and low nutrient content.

1.26.2 HISTORICAL EVIDENCE FOR EROSION

Human impact in the form of deforestation, grazing and agriculture was considerable in the Holocene, but especially acute in historic times. This combination of factors has been documented for several periods, but particularly well from the 16th to 18th centuries when cold, humid climatic fluctuations (Little Ice Age) coincided with land-use changes that left large areas exposed to erosion. The first half of the 20th century, when overpopulation of rural areas led to the expansion of agricultural land into rangelands, was another period of heavy erosion due to excessive use and high soil loss rates (Puigdefábregas, 1995).

A large part of Spain shows signs of erosion. However, not all erosive landscapes are of recent origin. As an example, in the desert of Tabernas (Figure 1.26.7), Almeria, the main drivers for erosion are not human

Figure 1.26.6 Aerial view of La Higueruela experimental station (Toledo province, central Spain), showing a mosaic of soil surfaces formed by erosion and deposition (photo from http://www.ccma.csic.es/). [Reproduced from www.ccma.csic.es with permission of Consejo Superior de Investigaciones Cientificas (CSIC)]

Figure 1.26.7 Erosive landscape not related to human activity: aerial view of well-developed badlands in the Tabernas desert, SE Spain (*photo*: A Solé-Benet)

activity and climate (although the driest in Spain) but tectonic activity since the begining of the Quaternary and the adjustment of the drainage network (Alexander *et al*., 1994), producing one of the most scenic badlands in Spain. A similar statement is true for the badlands of Guadix, which are about 4000 years old (Wise *et al*., 1982).

Geomorphological investigations carried out on fluvial terraces of Mediterranean rivers indicate that higher erosion and sedimentation rates prevailed in the last 2000 years (Marqués and Juliá, 1984) or even 500 years (Hoffman, 1987; Castro *et al*., 1998) (all cited by Grove and Rackham, 2001). The Andarax river, in south-eastern Spain, has a larger but very arid catchment, including the south-eastern slopes of Sierra Nevada, the northern part of Sierra de Gador and some of the largest badlands in Europe. The original Andarax estuary extended some 8 km above the present mouth (Figure 1.26.8). Boreholes have revealed marine and brackish-water lagoon sediments, just below present sea-level, representing an accumulation of about 1 m thickness per 1000 years. They are covered by coarse sediments brought down by floods within the last few millenia. A Punic sherd dating from about 600 BC was found in them at a depth of 2.4 m. Since the mid-eighteenth century, the river, owing to extensive deforestation for mining, has extended its delta by about 6 km^2 (Hoffman, 1987, cited by Grove and Rackham, 2001).

In the Ebro basin, Van Zuidam (1975) found a maximum period of erosion and alluviation between 500 and 100 BC, which he attributed to the growth of Celtic settlement and cultivation, especially of corn and wine. In the same basin, Gutiérrez Elorza and Peña-Monné (1998) found widespread slope accmulation between 900 and 300 BC, which they attributed to solifluction in a cool wet period. The two interpretations are not necessarily incompatible (Grove and Rackham, 2001).

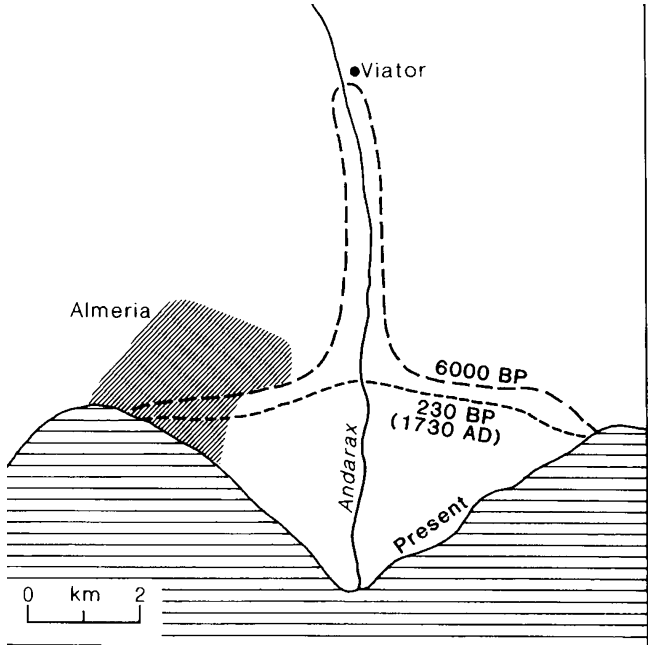

Figure 1.26.8 Map of the Andarax delta (Almeria, south-eastern Spain) showing the evolution of the coast line, according Marqués and Juliá (1984), in Grove and Rackham (2001). (Reproduced from Grove AT and Rackham O, *The Nature of Mediterranean Europe. An Ecological History*, Copyright 2001, Yale University Press)

Landscapes in many regions exhibit truncated soil profiles. Most organic horizons have been removed by erosion and B and C horizons are being cultivated. Aerial photographes of many parts of the country reveal a mosaic of coloured soil and pale subsoil, as a direct result of that process (Figure 1.26.6).

1.26.3 HOW EROSION IN SPAIN IS APPROACHED BY SCIENTISTS

The sudy of erosion has been undertaken by many reseach teams from Spanish and other European universities, from the Consejo Superior de Investigaciones Científicas (CSIC) or National Research Council, but also by the national and the regional autonomous administrations (Table 1.26.2).

Given the necessary synthetic characteristics of this chapter, it is imposible to summarize, even cite, all the studies which have been carried out on soil erosion in Spain. For this chapter, 615 specific documents have been consulted, 350 references from the author's database up to February 2003, and the rest obtained by both direct inquiries to about 50 colleagues and an Internet bibliographic search (OCLC Firstsearch Service).

This number of documents is smaller than the 850 references (Añó, personal communication, June 2003) maintained by the Centro de Investigaciones sobre Desertificación (CIDE) in Valencia, in the BIB-ERON database (Añó *et al*., 2000). The aim of BIB-ERON is to classify the large number of scientific and technical publications on water erosion in Spain and aims to be a useful tool for researchers through its availability on the Web in the near future. Classification descriptors include *state of the art*, *erosion methodologies*, *direct quantification and analysis of processes* to facilitate evaluation and retrieving. There are over 100 published documents from a recent (July 2003) symposium held in Madrid on 'Soil erosion and degradation control'.

The 615 documents consulted have mostly been published since 1980, as there are only 12 references before that year; 110 references were found between 1981 and 1990, 356 between 1991 and 2000 and 137 after 2000, indicating a geometric progression in publications on soil erosion in Spain in the last two decades. These observations agree with a larger survey of published work on geomorphology in Spain, where erosion was one of the topics considered (García-Ruíz, 1999).

During the last decade, the European Commission has contributed notably with the funding of several international research projects in which erosion was involved (MEDALUS, EPHEDA, CORINE, PESERA, etc.).

1.26.3.1 Summary of Techniques and Approaches Used in Spain

Gerlach troughs have been used for the assessment of sediment production from non-bounded plots in different environments, essentially on steep slopes and related to (a) forest fires (Soler and Sala, 1992), (b) reforestation techniques (Olarieta *et al*., 1999), (c) vegetation types (Sala and Calvo, 1990), and (d) vegetation density (Sánchez and Puidfefábregas, 1994).

Erosion pins and microprofiles techniques have been reviewed by Sancho *et al*. (1991) and roughness indices from microtopographic techniques have been related to water erosion by Vidal Vázquez *et al*. (1999).

Bounded plots of many sizes, including USLE ones, have been used by many (Diaz-Fierros *et al*., 1987; García Ruiz *et al*., 1995; Gutierrez Elorza *et al*., 1995; Nicolau, 1996; Rodriguez Martinez-Conde *et al*., 1996; Rubio *et al*., 1994; Bienes and Torcal Sáinz 1997; Kosmas *et al*., 1997; Bienes *et al*., 2001; Chirino *et al*., 2001; De Alba *et al*., 2001; Martínez-Mena *et al*., 2001, 2002).

Different simultaneous techniques have been used by Gutiérrez Elorza *et al*. (1995) and Sirvent *et al*. (1996, 1997), and they concluded that soil loss recorded using dynamic methods is lower than that recorded by erosion pins, and both underestimate the losses recorded with micro-topographic profiles.

Rainfall simulators either with sprinklers (Benito *et al*., 1986; Calvo-Cases *et al*., 1988, 1991; Navas *et al*., 1990; Quirantes *et al*., 1991 Cerdà *et al*., 1997; Solé-Benet *et al*., 1997, 2002) or drippers have been reported.

The assessment of rain erosivity and the prediction of rainfall events have been approached by ICONA (1988), García-Ruiz (2000) and Usón and Ramos (2001).

^{137}Cs has been used in the assessment of soil erosion and sedimentation (Navas and Machín, 1991; Navas and Walling, 1992; Quine *et al*., 1994).

Dendrochronology has been used as a method of assessment of gully erosion rates (Vandekerckhove *et al*., 2001), by datable deviations of normal growth pattern.

Many authors have used soil aggregate stability in the laboratory as an indicator for soil erodibility (Solé Benet *et al*, 1992; Josa *et al*., 1994; Cerdà, 1996, Ternan et al, 1996 Boix-Fayos *et al*, 2001; Lado *et al*, 2004).

Experimental weathering procedures have contributed to explaining the mechanisms of sediment delivery, especially in badland areas (Regüés *et al*. 1993, 1995; Pardini *et al*., 1995; Cantón *et al*., 2001).

Recently, spatial technology approaches coupled with remote sensing have been used for mapping and quantification of gully erosion (Martinez-Casasnovas, 2003; Ries and Marzolff, 2003; Vandekerckhove *et al*., 2003). Modelling of gully erosion has been undertaken by Casali *et al*. (2003) and even overall erosion (Del valle Melón *et al*., 1998).

Vandekerckhove *et al*. (1998, 2000) have studied, by means of direct measurements of geometric and topographical parameters coupled with photointerpretation, the characteristics and controlling factors of gullies and the thresholds for ephemeral gully initiation in south-eastern Spain. Kirkby *et al*. (2003) have observed and modelled the distribution of channel and gully heads in south-eastern Spain.

Verstraeten *et al*. (2003), from an existing dataset of area-specific sediment yield (SSY) for 60 catchments in Spain that was retrieved from sediment deposition rates in reservoirs (Avendaño Salas *et al*., 1997), found that the catchment area alone explains only 17% of the variability in SSY. The low prediction capability of the multiple regression models and the CORINE soil erosion risk map could be attributed mainly to the fact that

TABLE 1.26.2 Major contributions of scientists from Spanish and other European institutions to soil erosion

Institution	Institution or Department	Leader (and other members)	Location	Type of work[a]	Materials, methods[b]
Spain					
CSIC	ICTJA	F Gallart (D. Regüés, G Pardini)	Barcelona	a, b, c, e, j	A, B, C, D, L
CSIC	IPE	JM García Ruíz (T Lasanta, B Alvera)	Zaragoza	a, b, c, e, i	A, B, C
CSIC	CIDE	JL Rubio (V Andreu, C Añó) Valenciac	Valencia	c, e, d	A, B, C
CSIC	CEBAS	J Albaladejo (V Castillo, M Martínez)	Murcia	c, e, d, j	A, B, C
CSIC	EEZA	J Puigdefábregas (A Solé, M Boer, Y Cantón)	Almería	a, b, c, d, j	A, B, C, G, L
CSIC	CCMA	G Benito (S DeAlba)	Madrid	a, b, c	A,G
CSIC	EEZ	E Barahona (J Quirantes, L García)	Granada	a, b, c, e, f	D, L
CSIC	EEAD	A Navas (J Machin)	Zaragoza	e	J, K
CSIC	EEAD	JL Arrúe (MV Lopez)	Zaragoza	a, f, k	A
CSIC	IRNA	D de la Rosa	Sevilla	j	G, K
CEDEX	Erosion	L Avendaño (J Cobo)	Madrid	e	C, G, H
CEAM	Forest Restoration	RV Vallejo (S Bautista, E Chirino)	Valencia	d, j	A, B, C, F
DGCONA		L Rojo	Madrid	k	M
JUNTA ANDALUCIA		F Moreira	Sevilla		K
IMIA		R Bienes Allas	Madrid	e, k	A
U Barcelona	Geology	MA Marquès (E Mora)	Barcelona	a, b, c, d	A, L
U Barcelona	Geography	M Sala (R Batalla)	Barcelona	a, b, d	A, L
U Girona	Soil Science	G Pardini (G Dunjó)	Girona	a, c, e	A, F
UPC	Soil Science	JA Martínez-Casasnovas	Lleida	e	G, K
U Valencia	Geography	A Calvo (E Bochet)	Valencia	a, b, c, e, i, j	A, B, D, F, K
U Valencia	Geography	A Cerdá	Valencia	a, b, c, d, e, i	A,B, D,
U Alacant	Ecology	J Bellot (S Bautista, E Chirino)	Alicante		A, B, D, G
U Murcia	Geography	F López Bermúdez (MC Romero)	Murcia	a, b, c, e, i,	A, B, D, F,
U Cartagena	Soil Science	F Ortiz Silla (A Faz)	Murcia	a, e	
U Zaragoza	Geology	M Gutiérrez (G Desir, F Gutierrez)	Zaragoza	b, c, e, i	A, L
U Zaragoza	Geography	JL Peña (M Echeverría, JC González)	Zaragoza		
U Santiago	Soil Science	F Díaz Fierros (B Soto)	Santiago	a, b, c, d	A, B, C, D, L
U Coruña	Soil Science	A Paz (E Vidal, M Taboada, M Valcarcel)	A Coruña	e, j	L
U Santiago	Geography	R Rodriguez Martínez-Conde	Santiago	e	A
UPM	Soil Science	C Roquero	Madrid	e, j, k	A
U Madrid	Geology	A Pérez	Madrid	b, c	A, B, I
U Alcala	Ecology	JM Nicolau	Madrid	1	A, B
U Extremadura	Geography	S Schnabel	Caceres	a, c, e	A, B, D, K, L

Other European countries					
TU Braunsweig	Geography	H Rohdenburg	Germany	e, j	B, C, G
Silsoe College	Geography	RPC Morgan	UK	b, c, e, j, k	A, B, C, D, G, L
King's College	Geography	JB Thornes	UK	a, b, c, d, e, h, j	A, B, C, D, F, G, K, L
U Middlesex	Geography	A Harvey	UK	b, h, i	C, G, K
U Amsterdam	Geography	AC Imeson	Netherlands	a, b, c, d, e, i, j	A, B, C, D, E, F, G, K, L
	Geography	H Scoging	UK	e, i	D
U Liverpool	Geography	H Faulkner	UK	a, c, i	
U Leeds	Geography	MJ Kirkby	UK	e, j	G
UK Leuven	Geography	J Poesen *et al.*	Belgium		
Silsoe College	Geography	JN Quinton	UK	a, b, e, j	A, B, D, F, L
UC Louvain	Geography	B Van Wesemael	Belgium	c, e, g, j	A, B, D, F, K, L

[a] *Field of work (in relation to erosion studies)*:
a land abandonment and collapse of traditional structures
b natural extreme events
c processes
d burned vegetation or forest fires
e water erosion
f wind erosion
g tillage erosion
h mass movements
i piping
j degradation of soil properties
k soil conservation, tillage methods
l mine tailings

[b] *Materials and methods*:
A experimental plots
B micro-catchments
C catchments – basins
D rainfall simulations in the field
E rainfall simulations in the laboratory
F aggregate stability (laboratory)
G models
H sediments in reservoirs
I sedimentology
J ^{137}Cs
K mapping, GIS
L direct measurements (profiles, pins, etc.)
M revegetation techniques

these methods do not incorporate gully erosion and that the land-cover data are not a good representation of soil cover. Other authors have also analysed soil erosion through the silting-up of Spanish reservoirs (Schnabel and Ergenzinger, 1987; Navas *et al*., 1998).

In recent years, the 'Soil Evaluation Group' from IRNA–CSIC has gathered in a structured format the available information about the quality of Spanish soils, and its degradation state is available online (http://www.microleis.com) (Dela Rosa *et al*., 2001). MicroLEIS is an integrated system for land data transfer and agro-ecological land evaluation, with special reference to Mediterranean regions, and is available online in both Spanish and English versions. Within this system, *ImpelERO* (De la Rosa *et al*., 1999) is a hybrid model of expert decision trees and an artificial neural network to evaluate soil erosion processes and to predict soil loss by water erosion.

SURMODES (Puigdefábregas and Del Barrio, 2000) is a project that intends to set up an operational surveillance system for early warning of desertification risk at the country scale. The system is designed as a support to mitigation programmes and integrates social, economic and landscape factors in its diagnoses and forecasts. Up to the present, a surveillance system has been developed with four modules: (i) early warning of risk, (ii) long-term monitoring of land cover change, (iii) databases and (iv) an observatory network with six terminals in representative landscapes of the country, linked through a telemetry system that works in a non-centralized way through an Internet backbone.

In recent years, soil erosion research has been heading towards the development and/or application of event-based physical models which could be validated by erosion datasets. Several researchers have applied existing models under local or regional environmental conditions, e.g. EUROSEM and WEPP. (Albaladejo *et al*., 1994), and on the development of new ones (De la Rosa *et al*., 1999).

1.26.3.2 Factors, Triggering Mechanisms and/or Other Features Associated with Soil Erosion

Extreme rainfall events have been studied by Gallart and Clotet-Perarnau (1988), White *et al*. (1997), García-Ruíz (2000), Cantón *et al*. (2001), Martínez-Mena *et al*. (2001), among many others.

The influence of forest fires (Mangas *et al*., 1992; Marques and Mora, 1992, 1998; Soler and Sala, 1992, 1995; Soto *et al*., 1994; Ubeda and Sala, 1998; DeLuis *et al*., 2001, 2003; Giovannini *et al*., 2001; Perez Cabello *et al*., 2002), after the pioneering work of two teams, one from Galicia (Díaz-Fierros *et al*., 1982, 1987) and the other from Valencia (Sanroque *et al*., 1985). Soil restoration after forest fires has been studied by the CEAM team (Bautista *et al*., 1996; Sanchez, 1997; Chirino *et al*., 2001).

Vegetation density and its spatial distribution clearly affect sediment transport and sedimentation, as shown by Andreu *et al*. (1998), Puigdefábregas *et al*. (1996), Bellot *et al*. (1998), González-Hidalgo *et al*. (1999) and Chirino *et al*. (2001), Bochet *et al*. (2002). The effects of vegetation removal have been studied by Castillo *et al*. (1997), Albaladejo *et al*. (1998) and Solé-Benet *et al*. (2002) in semi-arid south-eastern Spain and by Benito *et al*. (2003) in Galicia.

Tillage affects the pattern of rock fragment cover (Poesen *et al*., 1997) and the effects of tillage with a mouldboard plough on erosion have been modelled by De Alba *et al*. (2001) and De Alba (2003). Minimum tillage effects on Mediterranean dry farming erosion have been estimated using the USLE: losses of over $200\,t\,ha^{-1}$ with conventional tillage to less than $20\,t\,ha^{-1}$ with mimimum tillage (Giraldez *et al*., 1989).

The abandonment of terraces consistently affects erosion (Arnáez *et al*., 1992; Llorens and Gallart, 1992; Marco Molina *et al*., 1996 Ruecker *et al*., 1998; Lasanta, 2001; Gisbert *et al*., 2002; Reynés *et al*., 2002; Dunjó *et al*., 2003) and the influence of soil management (De Alba *et al*., 2001). Land-use changes, especially in mountain areas (from forests to pastures and from pastures to ski resorts) greatly enhance erosion (Del Barrio and Puigdefábregas, 1987; García-Ruíz *et al*., 1990). Conversion of matorral to *Pinus* forest has been studied by Ternan *et al*. (1996) for its erosional impact.

Also the construction of new terraces, with heavy machinery, for forest revegetation purposes, has triggered water erosion in some regions (García-Pérez, 1999). Gully erosion is accelerated by runoff water shed from terrace treads (Ternan *et al*., 1996).

The effects of land reshaping mostly for agricultural purposes (Poesen and Hooke, 1997), but also for other purposes, such as ski resorts in the Pyrennees, enhance erosion and can result in mass movements or in significant increases in both surface and shallow subsurface runoff (Puigdefábregas and Alvera, 1986; García-Ruíz and Del Barrio, 1990).

The effects of crusting have been studied by Taboada Castro *et al*. (1999), Ramos *et al*. (2000) and Solé-Benet *et al*. (1997, 2002), among others.

One of the effects of mining activities, both underground and open-mine, is the removal of fertile soil, which, according to relatively recent laws, should be replaced or restored once the mining activity has ceased (Jorba *et al*., 2000). Erosion on mine tailings and on remodelled mine landscapes has been studied by Clotet *et al*. (1983), Porta *et al*. (1989), Wray (1998), Nicolau and Asensio (2000) and Nicolau (2003).

Motorway and railroad embankments have received little attention from an erosion point of view, despite the fact that they have often to be repaired and bear high associated costs (Andres and Jorba, 2000; Arnaez and Larrea, 1994). Also, roads and streets in new urban areas have had dramatic consequences for water erosion in some regions (Inbar and Sala, 1992).

The catastrophic failure of an earth dam built on gypsiferous alluvium and dispersive clays has been described (Gutierrez *et al*. 2003).

The causes of mass movements have been reviewed by Corominas (1989), and Araña *et al*. (1992) reviewed geological risks and factors, including erosion, in Spain.

Bedload transport in channels has been studied by Conesa García (1995), Batalla *et al.* (1995) and Rodriguez Martinez-Conde *et al*. (1998), among others.

1.26.4 CURRENT EROSION PROCESSES AND RATES

1.26.4.1 Water Erosion (Figures 1.26.9 and 1.26.10)

Water erosion is the most important type of soil erosion in Spain. Most severe problems occur where rainfall erosivity is high (most of the Mediterranean coast but increasing towards the north-east) and vegetation density is low (autumn is the season with the least protected agricultural soils). The main processes include splash, inter-rill, rill and gully erosion, river bank erosion and pipe erosion, although in general no quantification is provided for specific water erosion processes. Quantities are usually given for total erosion in specific gullies, or on plots, hillslopes and catchments in relation to land use, soil management, plant cover and forest fires.

1.26.4.1.1 Splash

Regüés *et al*. (1995) described the formation of pedestals as a consequence of rainfall experiments over mudstone regolith. A new type of splash cup was designed and tested by Molina and Llinares (1996). Bochet *et al*. (2002) measured in Valencia the influence of plant morphology on soil detachment and Downward (2000) in the Tabernas badlands described the splashed material produced by different types of soil crust under a variety of rainfall events.

1.26.4.1.2 Rill and Inter-rill

Rill and inter-rill erosion are the most commonly described and quantified processes based on direct measurements (profile-metres, erosion pins), experimental runoff plots and micro-catchments, with both

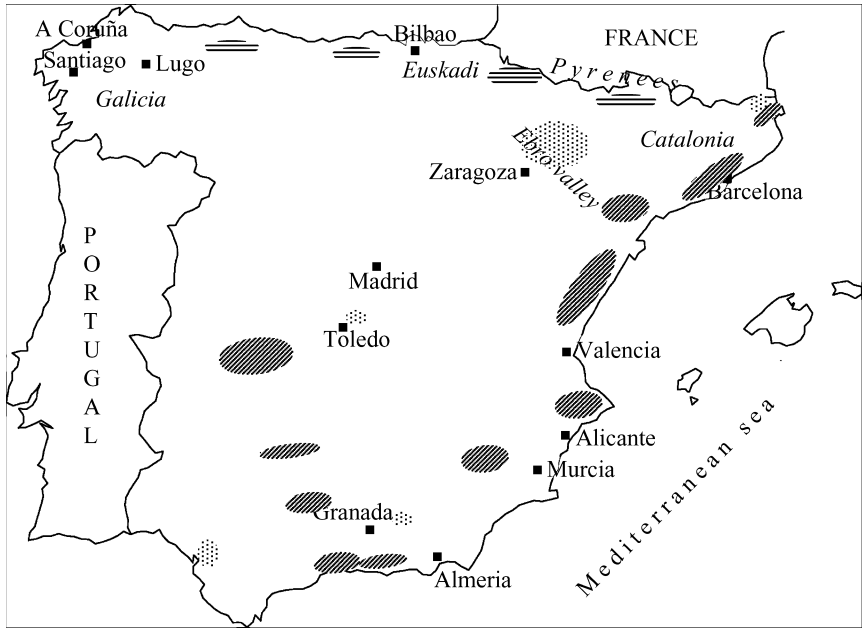

Figure 1.26.9 Hot spots for water erosion (areas with leaning lines), wind erosion (dotted areas) and mass movements (areas with horizontal lines)

natural and simulated rainfall. Cerdà (2001), in a review of soil erosion only in the Valencia autonomous region, provides data from 72 runoff plots and 109 rainfall simulations, with maximum values for plots of $18.46\,\mathrm{t\,ha^{-1}}$ on a $40 \times 8\,\mathrm{m}$ bare plot in a single event (Sánchez *et al.*, 1994), and $8.1\,\mathrm{t\,ha^{-1}\,yr^{-1}}$ under matorral and a total precipitation of 459 mm during one year (Table 1.26.3). For rainfall simulation,

Figure 1.26.10 Water erosion in a vineyard in Catalonia (*photo*: JA Martínez-Casasnovas; reproduced with permission)

maximum values of 26 t ha^{-1} h^{-1} under rainfall intensities of 55 mm h^{-1} have been reported on marls in badlands (Cerdà and García-Fayos, 1997). These values are similar to those reported for areas of cultivated soils, matorral and forests, and also from badlands, where rates can exceed 100 t ha^{-1} yr^{-1}. Romero *et al.* (1999), on plots 10×2 m over 3 years under 300 mm of annual rainfall, showed an order of magnitude increase in erosion from fallow or shrubland to cereals (Table 1.26.3). Martínez-Mena *et al.* (2002) recorded on large plots (328 and 759 m^2) from less than 2 to 12 t ha^{-1} in 3 years on shrubland with slopes of 23–35% with annual rainfall of 286 mm (Table 1.26.3). Cerdà (1997) measured sediment concentration in runoff from more than 1 to 200 g l^{-1}, the latter in badlands, but under relatively dense plant cover erosion is dramatically reduced and sediment concentrations are usually less than 1 g l^{-1}. Chirino *et al.* (2001) found on bounded erosion plots in open pine forest sediment losses of 0.020 t ha^{-1} yr^{-1} and one order of magnitude higher, 0.310 t ha^{-1} yr^{-1}, under alpha grass and 1.9 t ha^{-1} yr^{-1} on bare soil (Table 1.26.3). Plant cover reduced erosion by up to 98% and an asymptotic model was obtained for the estimation of annual soil losses for a precipitation range of 330.4 ± 105 mm : $y = a \times b^{\mathrm{pc}} \times c^{\mathrm{LAI}}$ (where pc = plant cover and LAI = leaf area index).

Experiments in Fuente Librilla, Murcia, showed that the soil type determines the hydrological response, regardless of rainfall intensisty, and also a time-dependent size distribution of the eroded material (decreasing coarse fractions and increasing fine fractions with runoff time) was observed (Martínez-Mena *et al.*, 2002).

1.26.4.1.3 Gully Erosion

Gully erosion has been described and measured in several locations (Schnabel and Gómez-Amelia, 1993; Faulkner, 1995; Martínez-Casasanovas *et al.*, 1998; Vandekerkhove *et al.*, 1998, 2000; Casali *et al.*, 1999, 2003; Nogueras *et al.*, 2000; Poesen *et al.*, 2002) and most such studies conclude that gullies are a dominant sediment source. In two locations in south-eastern Spain, on abandoned agricultural land, the contribution of permanent gullies to mean sediment production over a 10-year period equals 83% of total sediment produced by water erosion, with annual rates of 37.6 and 9.7 t ha^{-1} (Poesen *et al.*, 2002) (Table 1.26.4).

Poesen *et al.* (2002) indicate that: 'the importance of sediment production by gullies in drylands can be assessed when comparing mean sediment deposition rates in Spanish reservoirs with the sediment production by inter-rill and rill erosion measured on runoff plots. Mean sediment deposition rate over a period of 5–101 years (Avendaño Salas *et al.*, 1997) in Spanish reservoirs with corresponding catchments ranging between 31 and 16 952 km^2 equals to 4.4 t ha^{-1} yr^{-1} and can even go up to 10 t ha^{-1} yr^{-1} or more (López-Bermúdez, 1990; Romero-Díaz *et al.*, 1992; Avendaño Salas *et al.*, 1997) (Table 1.26.5). These figures are significantly higher than reported short- to medium-term rates of inter-rill and rill erosion in the Mediterranean as measured on runoff plots (Andreu *et al.*, 1998; Castillo, 1999; Puigdefábregas *et al.*, 1999; Romero-Diaz *et al.*, 1999; Cerda, 2001), i.e. less than or equal to 0.1 t ha^{-1} yr^{-1} for shrubland and olive groves, 0.2 t ha^{-1} yr^{-1} for wheat and 1.4 t ha^{-1} yr^{-1} for vines (Kosmas *et al.*, 1997)'.

Moreover, a recent study in the catchments of 22 Spanish reservoirs indicates that specific sediment yield increases when the frequency of gullies increases in the catchment (Konickx, 2000, cited by Poesen *et al.*, 2002).

The ephemeral gully erosion model (EGEM) (Woodward, 1999) did not show a good relationship between predicted and measured ephemeral gully cross-sections in south-eastern Portugal (Nachtergaele *et al.*, 2002), although good relationships were found in north-eastern Spain on vineyards (Meyer and Martinez-Casasnovas, 1999).

Piping may have an important role in the initiation and to a lesser extent, development of some gully systems such as bank gullies and gullies forming on badland areas in Mediterranean regions (Poesen *et al.*, 2002).

TABLE 1.26.3 Soil loss by sheet and rill erosion from selected runoff plots in Spain

Reference	Location	Plot size (m)	Period	Slope	Soil type	Annual precipitation (mm)	Land use	Soil loss
Scoging, 1982	Granada	30 × 6	10–75 to 10–76	15–35°	Silty regolith	n.a.	Badland	36.9–93.6 t ha^{-1}
Marquès *et al.*, 1987	NE Spain	6.4 m^2	1993	5°	Typic Xerochrept	528.5	Orchard (contour cultivation)	24 t ha^{-1} yr^{-1}
Marquès *et al.*, 1987	NE Spain	6.4 m^2	1994	5°	Typic Xerochrept	628.5	Orchard (contour cultivation)	3.58 t ha^{-1} yr^{-1}
Francis, 1990	Murcia	3 m^2	3 years	5.6–10.3°	Marly regosols	<300	Fallow terraced fields	0.8–5.3 t ha^{-1} yr^{-1}
Sirvent *et al.*, 1993	Central Ebro basin	57 m^2	07–91 to 05–92	5°	Miocene clays	320	Bare rangeland	74 t ha^{-1} yr^{-1}
Sirvent *et al.*, 1993	Central Ebro basin	24 m^2	07–91 to 05–92	23°	Miocene clays	320	Bare rangeland	112 to 180 t ha^{-1} yr^{-1}
Rubio *et al.*, 1993	Valencia	11–34 m^2	09–92 to 04–93	25–35%	Haplic Calcisol	241–497	Burned pine forest	0.11 to 4.34 t ha^{-1}
Quine *et al.*, 1994	Ebro basin	52 ha	n.a.[a]	5–15°	Rc - Bk[b]	450	Cultivated (cereals)	16–25 t ha^{-1} yr^{-1}
Quine *et al.*, 1994	Ebro basi	52 ha	n.a.	5–15°	Rc - Bk[b]	450	Shrubland	2–4 t ha^{-1}
Poesen *et al.*, 1996	Almeria	n.a.	1983–93	1–5°	Mollic Palexeralf	<200	Abandoned	2 m^3 ha^{-1} yr^{-1}
Cerdà, 1997	Valencia	0.25 m^2	One event	>10°	Marl	60 mm h^{-1}	Badland	21.98 t ha^{-1} h^{-1}
Vila García *et al.*, 1998	Galicia	25–50 m^2	04–96 to 12–96	5–10°	Humic Cambisol	850	Cultivated (20–90% plant cover)	18 t ha^{-1}
Vila García *et al.*, 1998	Galicia	25–50 m^2	1997	5–10°	Humic Cambisol	1845.6	Cultivated (50–90% plant cover)	13–18 t ha^{-1}
Romero *et al.*, 1999	El Ardal (SE Spain)	10 × 2	01–91 to 08–94	7–28%	Xerollic Paelorthid	300	Fallow	0.088
Romero *et al.*, 1999	El Ardal (SE Spain)	10 × 2	01–91 to 08–94	7–28%	Xerollic Paelorthid	300	Shrubs	0.0657
Romero *et al.*, 1999	El Ardal (SE Spain)	10 × 2	01–91 to 08–94	7–28%	Xerollic Paelorthid	300	Cereals	0.614
Romero *et al.*, 1999	El Ardal (SE Spain)	10 × 2	01–91 to 08–94	7–28%	Xerollic Paelorthid	300	Cleared shrubs	0.455
Edeso *et al.*, 1999	Euskadi	5 × 2	01–94 to 07–94	40–50%	Forest soil	1500	Three treatments[c]	39.69 t ha^{-1} h^{-1}

Edeso *et al.*, 1999	Euskadi	5 × 2	01–94 to 07–94	40–50 %	Forest soil	1500	Stem-only harvested	9.32 t ha^{-1} h^{-1}
Duiker *et al.*, 2001	SW Spain	0.75 × 1	1998	30 %	Vertisol	60 mm h^{-1}	Experim. field	4.15 t ha^{-1} h^{-1}
Duiker *et al.*, 2001	SW Spain	0.75 × 1	1998	30 %	Yellow Alfisol	60 mm h^{-1}	experim. field	12.03 t ha^{-1} h^{-1}
de Alba *et al.*, 2001	Central Spain	25 × 5	4 years (93–97)	9 %	Typic Haploxeralf	450	No tillage	3.6 t ha^{-1} yr^{-1}
de Alba *et al.*, 2001	Central Spain	25 × 5	4 years (93–97)	9 %	Typic Haploxeralf	450	Conventional tillage	6 t ha^{-1} yr^{-1}
de Alba *et al.*, 2001	Central Spain	25 × 5	4 years (93–97)	9 %	Typic Haploxeralf	450	Fallow	7.3 t ha^{-1} yr^{-1}
Ros *et al.*, 2001	Murcia	10 × 3	8 events	15 %	Xeric Torriorthent	175 (8 events)	Control plot	3 t ha^{-1} yr^{-1}
Ros *et al.*, 2001	Murcia	10 × 3	8 events	15 %	Xeric Torriorthent	175(8 events)	Sewage + compost amended	0.17 t ha^{-1} yr^{-1}
Martinez-Mena *et al.*, 2002	Murcia	2 × 1	Rainfall simul.	10–15 %	Calcaric regosol	30–60 (event)	Orchard	0.03 t ha^{-1} min^{-1}
Martinez-Mena *et al.*, 2002	Color (SE Spain)	328 m^2	26 events in 3 yr	35.50 %	Xeric Calcigypsid	286	Shrubland	1.89 t ha^{-1} in 3 yr
Martinez-Mena *et al.*, 2002	Abanilla	759 m^2	35 events in 3 yr	22.90 %	Xeric Torriorthent	286	Shrubland	12.07 t ha^{-1} in 3 yr
Benito et al. 2003	Galicia	1 m^2	One event 1998	19 %	Distrudepts	64 mm h^{-1}	Deforested land	3.28 t ha^{-1} h^{-1}
Benito et al. 2003	Galicia	1 m^2	One event 1999	24 %	Distrudepts	64 mm h^{-1}	Woodland	0.0094 t ha^{-1} h^{-1}
De Luis *et al.*, 2003	Alicante	2 × 2	One event	26–27°	Typic Calcixeroll	273 (event)	Shrubland	0.02–0.06 t ha^{-1} yr^{-1}
De Luis *et al.*, 2003	Alicante	2 × 2	One event	26–27°	Typic Calcixeroll	273 (event)	Burned shrubland	0.3–8.4 t ha^{-1}
Martinez-Raya, 2003	Granada	24 × 6	One event	33–38 %	Semi-arid soil	13.7 (event)	Traditional tillage	0.173 t ha^{-1}
Martinez-Raya, 2003	Granada	24 × 6	One event	33–38 %	Semi-arid soil	13.7 (event)	No tillage	0.331 t ha^{-1}
Martinez-Raya, 2003	Granada	24 × 6	One event	33–38 %	Semi-arid soil	13.7 (event)	Full plant cover	0 t ha^{-1}

[a] n.a., Not available.
[b] Rc = Calcaric Regosol; Bk = Calcic Cambisol.
[c] tree harvested + litter removed + ploughing.

TABLE 1.26.4 Soil loss from selected gullies in Spain[a]

Reference	Location	Size (ha)	Period	Slope (%)	Soil type	Annual precipitation (mm)	Land use	Soil loss[a]
Poesen *et al.*, 1996	Almeria		1983–93	n.a.	Mollic Palexeralf	<200	Abandoned	9.7 $m^3\ ha^{-1}\ yr^{-1}$
Schnabel *et al.*, 1998	Extremadura	35.4	7.5 years	n.a.	Brown Mediterranean–Lithosols	514.3	*Dehesa*	39.05 $m^3\ yr^{-1}$
Casali *et al.*, 1999	Navarra		2 years		Badland	Usual in the area	Badland	26.6 $t\ ha^{-1}$
Hooke and Mant, 2000	Murcia	4110	Sep-97	0.0072	Many types	365	Grazing	53.185 $kg\ m^{-1}$
Oostwoud Wijdenes *et al.*, 2000	Murcia		04–97 to 03–99		Quaternary fill and marls	300	Shrubs, alpha grass	4 $m^3\ yr^{-1}$
Martinez-Casasnovas *et al.*, 2002	Catalonia	2.12	One event	8.90	[b]	215 mm event	Cultivated	207 $t\ ha^{-1}$
Valcarcel *et al.*, 2003	Galicia	36.8	1997–99	15	Silt-loam and loam	1000–1500	Cultivated	2.12 $m^3\ ha^{-1}$

[a]Note that soil loss is expressed differently according the different studies (volume or weight per contributing area per year, weight per year or weight per gully length).
[b]Xerorthent typic, Calcixerept typic, Calcixerept petrocalcic, Haploxerept fluventic.

TABLE 1.26.5 Sediment yield from selected large basins and catchments in Spain

Reference	Location	Size (km^2)	Period	Annual precipitation (mm)	Land use	Soil loss ($t\,ha^{-1}\,yr^{-1}$)
Wise *et al*., 1982	Granada (SE)		2000 years	400	Badlands	0.16–0.4
Benito *et al*. 1991	Miño–Lugo (NW)	2303	1990?	n.a.	n.a.	4.7
Benito *et al*. 1991	Tambre-Portón (NW)	1146	1990?	n.a.	n.a.	9.3
Avendaño *et al*. 1997	Embarcaderos (SE)	18952	>10 years	600	n.a.	0.17
Avendaño *et al*. 1997	Riudecañas (NE)	31	>10 years	800	n.a.	1.12
Avendaño *et al*. 1997	Puentes (SE)	1042	101 years	400	n.a.	2.02
Avendaño *et al*. 1997	Guadalest (E)	60	>10	600	n.a.	27.03
Lajournade *et al*., 1998	Central Pyrennees	18.8	One event	160 mm in 2 h	Forest	$67\,t\,ha^{-1}$
Arnaez *et al*., 1998	Central Pyrennees	2.84	11 months	1140	Abandoned fields	$1.21\,t\,ha^{-1}$
Regüés *et al*., 1988	Eastern Pyrenees	4.16	1993–98	850	[a]	90–150
Regüés *et al*., 1988	Eastern Pyrenees	0.17	1989–98	850	[b]	0.19

[a] 43% forest, 21% terraces, 3% badlands.
[b] 55% abandoned terraces, 10% forest.

1.26.4.1.4 Pipe Erosion

Pipe erosion has also been studied at several locations where high-risk materials are abundant (Tertiary sedimentary basins): Harvey (1982) and Faulkner *et al.* (2000) in Almeria; López-Bermúdez and Torcal Sáinz (1986) and López-Bermúdez and Romero-Díaz (1989) in Murcia; Martín-Penela (1994) in Granada; García-Ruíz *et al*. (1997a) in the Central Pyrenees; Gutiérrez *et al*. (1997) in the Ebro valley.

1.26.4.2 Mass Movements

In Spain, owing to its geological, orographic and climatic characteristics, the risk of landslides on slopes is significant. Yearly losses due to damage by landslides is calculated to be over €120 million (Ayala *et al*., 1987), which, updated by means of the index of consumer prices (INE 2003), would now be around €240 million. The wide variety of lithologies, morphologies and climate zones in Spain causes irregular distribution of hillside instability phenomena. The western and central sectors of the country that make up the Hercynian base of the Meseta are the least problematic owing to the resistance of its materials (plutonic rock, gneiss, quartzite, schists) and gentle morphology. In contrast, the peripheral Alpine mountain ranges record the greatest number of phenomena (Figure 1.26.9), owing to their young relief, high rainfall and the presence of lithologies susceptible to mass movements. Corominas (1989) inventoried all lithologies suceptible to mass movements, indicating their location and the mechanism by which mudflows are produced (Corominas and Moreno, 1988).

Araña *et al*. (1992), in an exhaustive review of geological risks in Spain, list the main catastrophic landslides recorded since 1620, indicating the type of movement, volume eroded and their effects. The most outstanding are a $10^7\,m^3$ complex translational slide in Pont de Bar (Lleida) in November 1982, which destroyed the entire town and the road, a $3.6 \times 10^6\,m^3$ mudflow in Olivares (Granada) in April 1986, which partially destroyed the town, and another $10^6\,m^3$ mudflow in Inza (Navarra) between December 1714 and April 1715, which destroyed the town.

There are several risk maps at different scales, from 1:1 000 000 (IGME, 1987), in which the zones affected by the different types of mass movements are shown, and some regional 1:400 000 maps, to the 1:100 000

maps in which the MOPU Geological Service shows problematic areas, types of movements, susceptible lithological formations and angles of stability. The IGME also has made 1:25 000 geotechnicial and geological risk maps for 15 Spanish cities, showing mechanical characteristics of the ground, and there are 1:10 000 and 1:5000 maps showing landslide risk (Araña *et al*., 1992).

There is also an extensive bibliography of specific studies of phenomena related to landslides on hillsides, especially in mountain areas (Del Barrio and Puigdefabregas, 1987; García-Ruíz *et al*., 1990; Cendrero and Dramis, 1996; González-Diez *et al*., 1999), individual studies on the prevention of risks caused by instability, inventories of landslides (Corominas, 1989) and calculations of economic losses (Ayala *et al*., 1987).

1.26.4.3 Wind Erosion

Wind erosion has been reported only locally in susceptible areas (north-western and southern coastal areas, some spots in north-eastern Spain and in the middle Ebro valley) (Figure 1.26.9). After the pioneering work of Quirantes *et al*. (1989) in the south-east, in which a series of maps at 1:400 000 were produced within the LUCDEME project, a new concern about the influence of tillage operations on wind erosion is growing in areas affected by strong W–NW winds (local name *cierzo*), mainly due to the work of Arrue's team at the Aula Dei Institute in Zaragoza. This latter has shown how in the semi-arid drylands of the middle Ebro valley, reduced tillage produces larger soil aggregates, greater surface roughness and more protective cover (by plant residues, aggregates and rock fragments), greatly decreasing the risk of wind erosion compared with traditional soil tillage (Lopez, 1998; Lopez *et al*., 1998, 2000, 2001; Sterk *et al*., 1999; Gomes *et al*., 2003) (Table 1.26.6).

1.26.4.4 Tillage Erosion

Tillage erosion has only recently received some notice (Poesen *et al*., 1997; De Alba, 1998; Quine *et al*., 1999). Poesen and Quine worked in the Guadalentin basin (south-eastern Spain), where a direct erosion displacement was given per pass, with the use of metal tracers. De Alba (1998), in Central Spain, with similar tracers, determined that tillage erosion was one order of magnitude larger than water erosion on plots with 15–30% slopes (54.7 and 7.3 t ha^{-1}, respectively) (Table 1.26.7).

However, more authors have studied the effects of different tillage methods on water erosion, such as De Alba *et al*. (2001) and De Alba (2003) in Central Spain and Valcárcel *et al*. (2002) in Galicia with the use of GIS to model the effect of agricultural factors such as the rotation scheme and the characteristics of the tillage system on surface water runoff and erosion. Also, Martínez-Raya *et al*. (2002) in Granada have evaluated

TABLE 1.26.6 Wind erosion rates in Spain from selected studies

Reference	Location	Plot size (m)	Period	Slope	Soil type	Annual precipitation (mm)	Land use	Soil loss (t ha^{-1} yr^{-1})
Sterk *et al*., 1999	Middle Ebro valley	135 × 180	1996–97	Level	Silt loam	365	Conventional tillage	0.5
Sterk *et al*., 1999	Middle Ebro valley	135 × 180	1996–97	Level	Silt loam	365	Reduced tillage	0
Lopez., 2001	Middle Ebro valley	135 × 180	17 months	Level	Sandy	<400	Fallow, mouldboard ploughed	>20
Ries *et al*., 2000	Middle Ebro valley	wind tunnel	Event	Level		n.a.	Fallow land	0.002–0.018

TABLE 1.26.7 Soil loss by tillage erosion in Spain

Reference	Location	Plot size (m)	Period	Slope	Soil type	Annual precipitation (mm)	Land use	Treatment	Soil loss ($t\,ha^{-1}\,yr^{-1}$)
Poesen *et al*., 1997	SE Spain	50 (length)		20%	Eutric Regosol	274	Almonds	Up-and-down tillage	54–88
Poesen *et al*., 1997	SE Spain	50 (length)		20%	Eutric Regosol	274	Almonds	Countour tillage	22–39
De Alba, 1998	Toledo	4.5 × 2.75	1995–96	9%	Calcic Luvisol	n.a.	Experimental	Tillage along slope	57.4
De Alba, 1998	Toledo	4.5 × 2.75	1995–96	9%	Calcic Luvisol	n.a.	Experimental	Contour tillage	5.9
Quine *et al*., 1999	SE Spain	n.a.	Pass	24°	[a]	275	Almonds	Conventional tillage	$200\,kg\,m^{-1}$ per pass
Quine *et al*., 1999	SE Spain	n.a.	Pass	<14°	[a]	275	Almonds	Duckfoot chisel	$657\,kg\,m^{-1}$ per pass
Martinez-Raya *et al*., 2002	Granada	n.a	1 event	>30 %	n.a.	17.2	Olives, almonds	Conventional tillage	0.17
Martinez-Raya *et al*., 2002	Granada	n.a	1 event	>30 %	n.a.	17.2	Olives, almonds	No tillage	0.068
Martinez-Raya *et al*., 2002	Granada	n.a	19 events	>30 %	n.a.	56.9	Olives, almonds	Legum as cover crop	5.0613
Martinez-Raya *et al*., 2002	Granada	n.a	19 events	>30 %	n.a.	56.9	Olives, almonds	Cereal as cover crop	3.6304

[a]Eutric Regosols and Calcaric Cambisols.

different tillage methods on steep slopes. López *et al*. (2003) have studied in dryland systems the impact of soil management on soil resilience and erosion.

1.26.5 MAJOR ON- AND OFF-SITE PROBLEMS AND COSTS

1.26.5.1 On-site Effects

According to data of the Directorate General for Nature Conservation (DGCONA), 48% of Spanish territory (220 000 km^2) shows a soil loss higher than soil tolerance (12 t ha^{-1} yr^{-1}) and 90 000 km^2 (18% of the total) is affected by very intense erosion rates higher than 50 t ha^{-1} yr^{-1}. The soil erosion affected areas are predominantly located in the Mediterranean basin.

A major consequences of soil erosion is reservoir siltation and this is reviewed in Section 1.26.4.1.3.

The abandonment of traditional land-use systems results in a loss of pastoral quality, soil erosion, fire risk and a decrease in biodiversity and threatens vulnerable species (González Bernáldez, 1991).

The protective role of forests on soils includes the maintenance of biological functions, the regulation of nutrients and the storage of carbon.. Martinez-Mena *et al*. (2002) have shown on experimental plots in Sierra de Orihuela that organic carbon decreased from 4 to 2.8% in the 9 years after vegetation removal. The carbon decrease is equivalent to an estimated loss of 46.8 t ha^{-1} of organic carbon, which is attributed to enhanced mineralization and oxidation of organic matter due to an increase in radiation and the temperature of surface soil layers (Martínez-Mena *et al*., 2002).

Soil erosion by water causes not only the loss of mineral components but also the loss of the organic fraction (organic matter, litter, etc.) and seeds, which are very important for the evolution of soils and landscapes (Cerdà and García-Fayos, 2002). In studies of the process of erosion of seeds, it was found that the interaction between vegetation and erosion that occurs at hillslope scale (e.g. and Puigdefábregas and Sanches, 1996) also occurs on a millimetric scale with seeds. Shapes, sizes, appendages and mucillage of seeds interfere in the erosion process determining the removal and deposit of seeds.

1.26.5.2 Off-site Effects

One of the most dramatic off-site effects of water erosion is that related to floods: morphological impacts and their relation to magnitude and frequency of floods in ephemeral streams of Mediterranean Spain (Lopez-Bermúdez *et al*., 2002). Such impacts include bank erosion, modifications of the channel where banks were overtopped, and floodplain sedimentation. In the 1973 flood on the Nogalte rambla in south-eastern Spain, sediment loads of 40% of the volume of flow (which reached over 2000 m^3 s^{-1}) were recorded (Heras, 1973, in Lopez-Bermúdez *et al*., 2002), resulting in many casualties and damage to buildings and civil works. However, this is only one case of 2400 recorded flood events in Mediterranean Spain since 1450 (Lopez-Bermúdez *et al*., 2002). Peak flow discharges over 1000 m^3 s^{-1} have been estimated for six Southern Spanish ephemeral rivers for a return period of 25 years (Heras, 1973, in López-Bermúdez *et al*., 2002).

Atlantic rivers, such as the Tagus, also produce important floods, estimated from historical documents or evaluated by means of paleohydrological methods (Benito, 2002; Benito *et al.*, 2003).

Another consequence of soil erosion is reservoir silting. The mean sediment deposition rate over a period of 5–101 years (Avendaño Salas *et al*., 1997) in Spanish reservoirs with corresponding catchments ranging between 31 and 16 952 km^2 equals 4.4 t ha^{-1} yr^{-1} and can even go up to 10 t ha^{-1} yr^{-1} or more (Avendaño Salas *et al*., 1997; Lopez Bermúdez, 1990; Romero-Diaz *et al*., 1992).

According to Olcina (1994), between 1983 and 1993, the economic losses caused by natural disasters in Spain, including earthquakes, never exceeded 1% of the gross national product, i.e. €3000 million at that time.

Taking into account that natural disasters include droughts, floods, mass movements, earthquakes, forest fires and soil erosion, is not easy to assign a given percentage to losses related to soil erosion. However, Ayala *et al*. (1988), estimated the potential losses for soil erosion during the period 1986–2016 at €5200 million (assessed in 1986), i.e. about €173 million per year, while landslides would cost between €5350 and 4500 million (assessed in 1986).

1.26.6 SOIL CONSERVATION AND POLICIES TO COMBAT EROSION AND OFF-SITE PROBLEMS

Chapter 2.23 by Fullen *et al*. deals with the same topic, so only complementary information is provided here.

Since the end of the 18th century, a few authors have shown their concern about erosion in Spain, and even considered it as one of the most important problems (Mallada, 1890). Specific research into the problem, however, did not start until the second half of the 20th century. At that stage, erosion was approached as a technical problem, and research was focused on the development of measures to avoid both sedimentation in reservoirs and damage to civil works. In 1955, the *Servicio Central de Conservación de Suelos* was created, but the first quantification of erosion was not available until the 1970s. At present, erosion is considered by different institutions within the Ministry of Environment (Dirección General de Conservación de la Naturaleza, formerly ICONA, and the 10 *Confederaciones Hidrográficas* or Basin Authorities).

According to an official report (Presidencia del Gobierno, 1977), most of the country was affected by severe water erosion, and only the north-western and northern-central regions were affected to a moderate degree. In 1987, ICONA, CSIC and some Universities, established the ongoing project LUCDEME (*Lucha contra la Desertificación en el Mediterráneo*) to combat desertification in Mediterranean drainage basins (Figure 1.26.11). Since then, a series of maps of actual and potential soil erosion have been produced.

The Spanish Forest Administration has long experience in protecting soil against water erosion and restoring degraded vegetable cover. Since 1901, when the Hydrology and Forest Divisions were created to revegetate thousands of hectares, several reforestation plans have been launched. From 1940 to 1980, more than 2.5×10^6 ha were afforested and complementary programmes for soil conservation and soil agricultural productivity maintenance were implemented. In the last decade, most responsibility for forest resources and nature conservation has been transferred to the Autonomous Communities from the Environment Ministry, although Central Government continues to coordinate plans and programmes related to soil protection and desertification control through DGCN (formerly ICONA, General Directorate for Nature Conservation).

However, the negative impact of some political measures on soil erosion, at regional, national and European scales, have been raised by Faulkner (1995), García Pérez *et al*. (1995) and García Pérez (1999), who mentioned very damaging soil preparation methods and the almost exclusive use of coniferous trees, among others. However, recent studies (Rojo Serrano *et al*., 2002) prove that mechanized afforestation techniques in the Guadalentin basin (south-eastern Spain), such as terracing and subsoiling, have been more effective than manual methods (holes, bench terraces and strips) in cutting hillslope runoff and retaining and storing as much water as possible.

Moreover, the rate of implementation of recent revegetation plans has been too slow to reverse erosion trends, and efforts to push back desertification should be stepped up (OECD conclusions and recommendations, 1997).

IN 1995, a network of experimental stations for monitoring and assessing erosion and desertification (RESEL) was established consisting of 47 representative field sites in problematic environments where erosion is being monitored at small scales on plots, hillslopes and/or in small catchments (Rojo Serrano and Sanchez Fuster, 1996). The RESEL network was formed by experimental stations from CSIC and some Universities. However, the scarcity of funding is a threat to its continuity.

In 2002, DGCN started a new national inventory of soil erosion (INES) with a 10-year periodicity with objectives to locate, quantify and analyse the evolution of erosion processes in Spain, with a final aim of giving

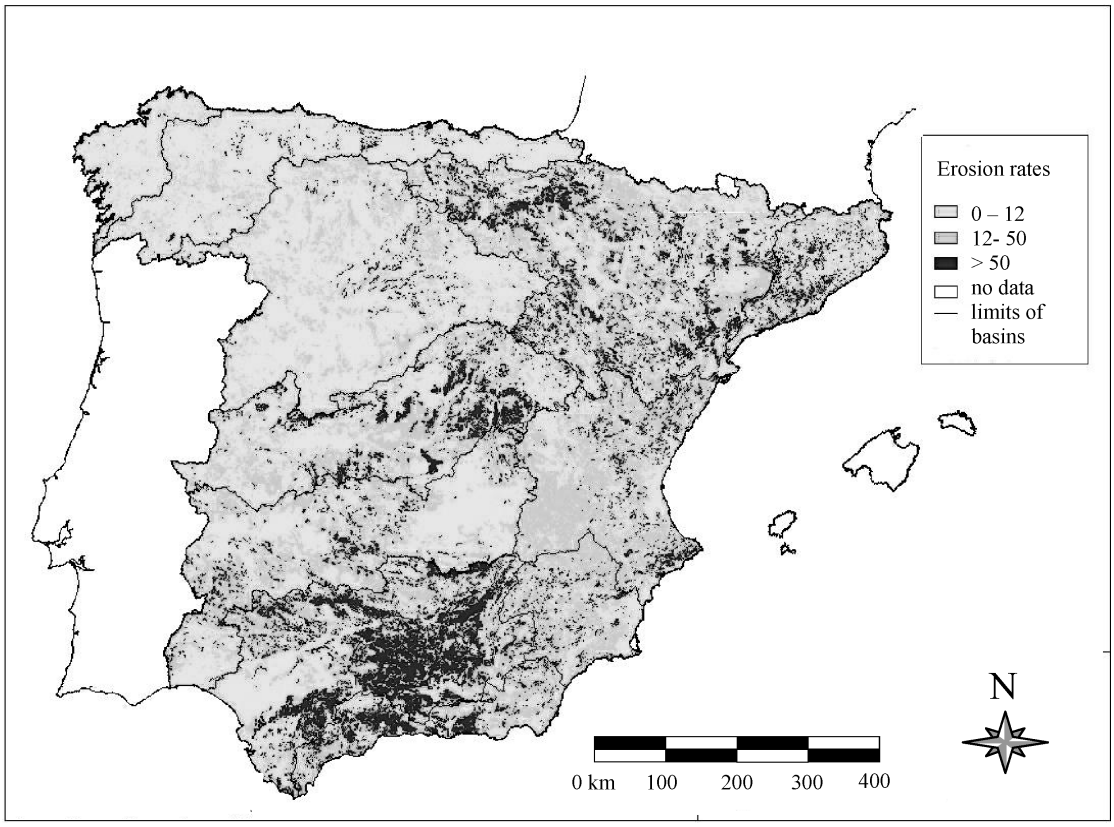

Figure 1.26.11 Erosion rates according to LUCDEME (1987). From lightest to darkest colour, erosion rates are 0–12, 12–50, $> 50\,t\,ha^{-1}$ (totally white areas have no data). (Reproduced from *Map of Soil Erosion in Spain*, 1987, with permission of LUCDEME)

priority to areas in which to fight erosion, and also to define and evaluate actions to carry out within the different national plans (reforestation, plant cover improvement and management of biodiversity in forests). For every province the following erosion types are inventoried and mapped (at a scale of 1:50 000): rill erosion, gully erosion, river bank erosion, mass movements and wind erosion. So far three provinces (Madrid, Murcia and Lugo) have been completed, five more are fairly advanced and other five are under way.

In addition to the national involvement in the assessment of soil erosion, local, regional and international concerns have been addressed by several organizations, but the results do not always agree. Sanchez Diaz *et al.* (2001) showed the discrepancies in some cartographic documents from ICONA (national), CORINE (European) and GLASOD (international), which might be due to different methodologies, input data and scales used.

1.26.7 CONCLUSIONS

In Spain, erosion is produced as a result of a set of processes over a variety of landscapes (forming a finer mosaic than in more humid areas). Centuries of anthropogenic action, especially in the Mediterranean region, have resulted in large areas of highly erodible, shallow soils with low organic matter content. Land-use changes and disturbances (urbanization, road construction, forest fires, abandonment of land, especially

terraces) have been reported as the main causes of severe erosion. Even reforestation of sensitive deforested areas has also been described as causing significant erosion.

Many of the reviewed documents indicate that accelerated erosion is a widespread and important concern in Spain and most emphasize the role of extreme events in long-term soil loss, especially in semi-arid regions.

What was stated by Wise *et al*. (1982) for south-eastern Spain regarding '. . .the difficulty of establishing contemporary rates of erosion: events are not only of high magnitude and infrequent occurrence, but also spatially discontinuous and greatly influenced by human activities' applies to most of Spain: erosion is more a collection of individual, local problems than a general one, as is commonly considered. Moreover, as most present erosion rates have been obtained from measurements on single gullies, small plots or small catchments, quantitative assessments of large areas should not be made by extrapolation. This effect of scale in erosion rates is extremely important: runoff is generated discontinuously on slopes so that fluxes of water transporting sediment from the top to the bottom rarely exist except in badlands, artificial taluses, roads, highways and urban zones. Sediments undergo a constant redistribution process in which plants play a fundamental role. Therefore, erosion is a slow process, although it can be accelerated under extreme events.

In spite of the initial alarm because of the high erosion rates estimated by the USLE, after 20 years of studies in Spain, it has been confirmed that, although there are erosion problems, severe erosion is restricted in space (specific areas of the country such as badlands, highway earthworks and restored zones) and in time (after fires, after agricultural abandonment, after ploughing). However, this does not mean that a broader perspective should not be considered in addressing soil erosion over the whole of Spain: erosion should be considered for a broad range of landscapes (steep and flat land) and relationships established for different land uses and management practices.

Erosion from rills and ephemeral gullies is more important than inter-rill erosion. At the agricultural plot scale, erosion along plot discontinuities (drainage paths, pathways, plot boundaries) and natural drainage pathways are much more important than erosion within plots, where most sediments remain. At the catchment scale, effective areas of sediment production are only a small percentage of the total catchment area.

Zones with intense natural erosion represent only a small loss of the overall soil resource, and may produce forms of high aesthetic value (especially in humid or sub-humid mountain regions), although they represent an important sediment source, which is its main nuisance from an environmental point of view (degradation of water quality, silting of reservoirs).

Soil conservation and protection measures should be applied following specific criteria for every region, taking into account physical and socio-economic factors, and considering spatial and temporal scales (recurrent torrential storms and droughts). The magnitude of soil loss tolerance for different environments and the capacity of such environments to withstand different soil losses should also be considered (a loss of 20 cm of soil over hard limestone is not comparable to the loss of a similar soil thickness over a soft parent material which is several metres thick).

In the years to come, soil erosion should be approached by modelling in which temporal and spatial scales are taken into account.

Finally, the study of soil erosion should not be dissociated from the essential study of Spanish soils (precise characterisation, formation processes and behaviour under different land uses and managements) or from the present and potential uses of the best soils, especially those from coastal areas which are being sealed by urbanization and roads. Regional characterization allowing soil conservation and a sustainable soil use should be a priority.

List of Abbreviations

CSIC	Spanish Research Council
CEAM	Centre for Environmental Mediterranean Studies
ICTJA	Earth Science Institute 'Jaume Almera'

EEZA	Experimental Station for Arid Zone Studies
EEZ	Experimental Station 'El Zaidin'
EEAD	Experimental Station 'Aula Dei'
IPE	Institute of Pyrenean Ecology
CIDE	Centre for Desertification Studies
CEBAS	Centre of Soil Science and Applied Biology of the Segura Basin
CCMA	Centre for Environmental Studies
UPC	Polytechnic University of Catalonia
UPM	Polytechnic University of Madrid
CEDEX	Centre for Experimental Studies on Civil Engineering
DGCN	Directorate General of Nature Conservation

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1.27

Spain: Canary Islands*

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1.27.1 INTRODUCTION

The Canary archipelago comprises a line of islands of volcanic origin 500 km long, occupying an area in the north-east of the Central Atlantic of approximately 100 000 km^2, near to the north-eastern African coastline, from which it is separated by a strip of sea 100 km wide. It lies, therefore, in a subtropical location, between latitudes 27° 37′ and 29° 25′ north and longitudes 13° 20′ and 18° 10′ west of Greenwich (Figure 1.27.1). The group of islands occupies 7447 km^2 in seven main islands and several islets. In order of decreasing size, these are Tenerife (2034 km^2), Fuerteventura (1660 km^2), Gran Canaria (1560 km^2), Lanzarote (846 km^2), La Palma (708 km^2), La Gomera (370 km^2) and El Hierro (269 km^2).

1.27.1.1 Climate

The Canary Islands occupy an area transitional between temperate and tropical regions. However, zonal geographic factors such as subtropical latitude, the proximity of the African continent, the cold Canary sea current, the trade winds and regional conditioning factors such as the very different relief of the islands and orientation relative to trade winds in the same island give rise to a large number of climatic factors that can be given the overall name of Mediterranean macrobioclimate (ESB, 1999).

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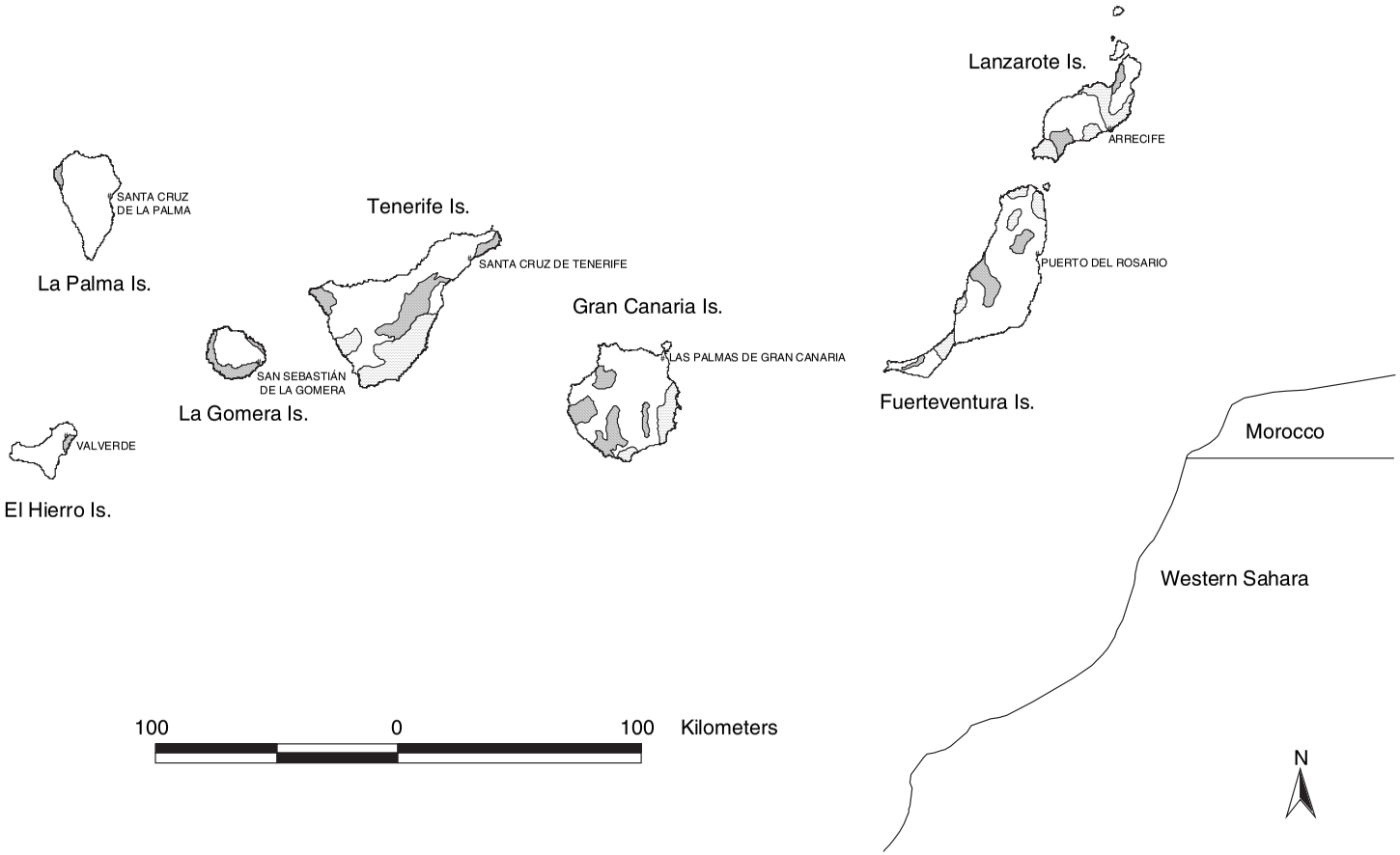

Figure 1.27.1 Water and wind erosion distribution in the Canary Islands (water erosion, dark shading; wind erosion, light shading)

As we move away from the African continent or ascend in altitude in the largest islands (Tenerife, La Palma, Gran Canaria, La Gomera and El Hierro), we can find bioclimates that range from desertic Mediterranean, arid and semiarid to mesophytic Mediterranean humid and subhumid. These give rise to soil moisture regimes that range from aridic to udic and xeric at altitudes higher than 1800 m.

The rainfall maximum is in winter and at the end of autumn (November, December, January) and the minimum in summer (July, August) and is highly variable in relation to altitude and orientation: from less than 100 mm yr^{-1} in the oriental islands and leeward coastal regions to more than 1100 mm yr^{-1} on the windward side of the central islands of greatest relief.

Erosivity of rainfall is also extremely variable in relation to geographic and atmospheric factors and the maximum intensity of rainfall ranges from 40 to 80 mm h^{-1} in sporadic events in intermediate and high windward areas of the islands with the greatest relief and between 60 and 100 mm h^{-1} in the eastern islands and leeward coastal zones where, although the total annual rainfall is low, the rains are intense and sometimes the total annual rainfall is concentrated in only one or two events.

The islands, especially the eastern ones, with the lowest height above sea level, are subject to northerly and north-easterly winds (55–80 % of all wind events), especially in the driest months (April to September) and mean speeds ranging from 6.3 to 7.0 m s^{-1}, resulting in a high erosive potential in seasons when the soil surface is completely dry and without plant cover.

1.27.1.2 Geological Surface Materials and Soils

The Canary Islands all have a volcanic origin; their formation began in the middle of the Tertiary and is still continuing. The geological materials are from subaerial volcanism and are comprised of basalts and to a lesser extent their salic differentiates (trachibasalts, phonolites and trachites), in the form of lava flows and pyroclasts.

The surface rocks are all of Pleistocene age (1.5–2.0 million years), although there are also Miocenic and Pliocenic outcrops in most of the islands and historical volcanic activity in the Islands of Tenerife, La Palma, Lanzarote and El Hierro from 1500 to the last eruption in 1971 (Teneguia volcano in La Palma).

This variability in the physical and lithological nature and in the age of the geological materials and the climatic conditions results in a very wide variety of soils.

Owing, in some cases, to the uneven topography and the steep slopes and, in others, to the young character of the geological material, predominant soils on the islands are Leptosols, Regosols and leptic and lithic subunits of other soils (25.3 % of the total surface of the archipelago) (Rodriguez Rodriguez *et al*., 2001a).

In the flatter areas, with older geological material, the dominant soil types are different in the wetter islands than in the more arid islands. The former are dominated by Andosols and andic subunits, Cambisols and Luvisols with small nuclei of Umbrisols, Acrisols and Vertisols, whereas in the more arid zones Calcisols and Solonchaks predominate with zones of Arenosols, Solonetz and Gypsisols.

The susceptibility of these soils to erosion is also very varied and we can again distinguish between wetter island soils, mostly of an andic nature, with a high organic matter content and a favourable structure, where erodibility values estimated from the *K* factor of USLE oscillate between 0.12 and 0.19 t yr^{-1} MJ^{-1} mm^{-1}, and soils of the most arid zones. In the latter, the low organic matter content and the existence of a high degree of salinity, and sometimes sodicity and a low degree of aggregation, make these soils less resistant to both water and wind erosion, with *K* factors between 0.27 and 0.34 t yr^{-1} MJ^{-1} mm^{-1} (Ortega *et al*., 1992).

1.27.1.3 Relief

There are two groups of islands. On the one hand, more arid ones of lower altitude (Lanzarote and Fuerteventura) have gentle hills and plains with scattered volcanic cones on which around 90% of the surface

has slopes of less than 12° and more than 50% less than 3°. The other group is comprised of the central and western islands, with steep cliffs in which slopes of over 12° occupy between 40 and 70% of the land and flat areas (<12°) are only found on the leeward coast in the two larger islands (Tenerife and Gran Canaria).

1.27.1.4 Vegetation and Land Use

The natural vegetation of the islands has been considerably transformed by intense human pressure since their conquest in the 15th century. This transformation has been especially intense since 1950 with the tourist boom, such that the islands are currently overpopulated, with 1 800 000 inhabitants in addition to the 12 000 000 tourists who visit the islands annually.

This results in a population density of 260 inhabitants km^{-2}, with maximum densities of 517 inhabitants km^{-2} recorded on Gran Canaria. If this figure is adjusted to the surface area of the islands usable and used by humans, the densities are almost of urban magnitude (a mean of 635 inhabitants km^{-2}, and 1600 inhabitants km^{-2} in Gran Canaria).

Natural vegetation of the islands is distributed into altitudinal bioclimatic levels following a gradient of increasing rainfall and decreasing temperature with increasing altitude.

In coastal areas with altitude below 350 m and in almost the whole of the lower islands with less relief, the climax vegetation is 'tabaibal–cardonal' (infra-mediterranean desertic arid series of the sweet tabaiba, *Ceropegio fuscae-Euphorbieto balsamiferae* S., and infra-mediterranean xerophytic lower semiarid series of the cardoon, *Periploco-Euphorbieto canariensis* S.).

However, this characteristic vegetation is only conserved if some enclaves are difficult to gain access to or are located in legally protected areas (40.4% of the surface area of the archipelago is legally considered as a Conservation Area). This is because these areas have suffered the most intense transformation owing to human activities being the main areas for tourist settlements (housing estates) and intensive irrigated agricultural production for export (greenhouses). Approximately 30% of the islands' population live at altitudes below 400 m, forming an urban continuum in these areas.

Even in the best conserved of these enclaves of plant communities, proportions of soil cover do not exceed 30–40% giving these areas little plant protection against aggressive climatic agents.

The land between 350 and 600 m is where traditionally, before tourist development, most agricultural and cattle farming activities were carried out by the islands' inhabitants and where most of the population lived before the main economic activity moved to coastal areas. This, therefore, corresponds to a completely transformed, although not degraded, zone, and the climax woodland (infra-thermomediterranean xerophytic upper semiarid series of the canarian savine, *Junipero canariensis-Oleeto* S.) has practically disappeared and has been replaced by a peculiar agrarian landscape of terraces, respectful of the environment and conservation of soils and water.

This is a space carefully managed by the farmer by sustainable methods and the abandonment of these traditional farming methods, owing to the attraction of economic activity in coastal areas, is resulting in deterioration of soil quality and a higher incidence of erosive processes.

In the north of the higher islands, between 600 and 1300 m, under the influence of trade winds from the north-east favouring the formation of cloud banks, laurel forest and 'fayal brezal' woodland are formed under the generic name of green forest (thermo-mesomediterranean mesophytic subhumid–humid series of the viñátigo, *Lauro-Perseeto indicae* S.). Traditionally, this woodland was exploited extensively by the inhabitants of neighbouring areas but is currently under complete environmental protection and can only be used for recreational purposes. These are woodland formations with a soil cover of almost 100% in which the soil erosion processes are greatly slowed because of the nature of the soils (Andosols).

Areas above 1300–1500 m, on the highest islands, feature pine forests (mesomediterranean mesophytic dry–lower subhumid series of the canarian pine, *Sideritido-Pineto canariensis* S.). These formations were

traditionally highly exploited for forestry purposes and also for extensive grazing and were also subject to frequent and repeated forest fires. However, they are now almost completely under legal protection and cannot be intensively exploited.

Pine forests are very open formations (50–60% soil cover) and the soils are frequently subject to water erosion processes in the form of rills and gullies.

Above 1500 m on the highest islands (Tenerife and La Palma), there are species-poor formations (hilltop shrubs) (supra-mediterranean mesophytic dry series of the Teide broom, *Spartocytiseto nubigeni* S.), in protected areas without any kind of human activity.

1.27.2 HISTORICAL EVIDENCE FOR EROSION

Until the islands' conquest by the Spanish (1450–1500), their inhabitants were mainly nomads and hunters whose activities had little impact on the land, which they maintained in fragile equilibrium with environmental conditions (Serra Rafols, 1986).

After this, the population's needs for agricultural land and grazing resulted in the disappearance of large areas of woodland and shrubland, breaking the climax equilibrium and producing intensive erosive processes, resulting in the silting up of some of the islands' natural lakes, giving rise to flood plain sedimentation, which occurred in La Laguna in Tenerife, or the formation of pediments, abundant in many of the islands, especially in Lanzarote and Fuerteventura.

This deforestation process was intensified around the middle of the 19th century, because of the growing needs for wood for the newly developing sugar industry, resulting in the near disappearance of the organic soil horizons from these areas, producing truncated profiles that are still visible today.

At the beginning of the 20th century, the population growth and a subsistence economy resulted in an increase in the area dedicated to crops. However, given the farmers' survival instinct and the inherent sustainability of their methods, they developed numerous structures to conserve soil and water. These consisted of large surface areas occupied by terraces, which today are considered as a most important natural, cultural and landscape feature that has helped to conserve the thin soils in the central islands.

With the change in the socio-economic model towards tourism and services after 1960–70, the islanders began to abandon this kind of agriculture and also, therefore, to abandon maintenance of the soil conservation structures, resulting in their deterioration and an acceleration of erosion processes.

This is demonstrated by the silting up of numerous reservoirs built in the middle of the 20th century. For example, Las Peñitas reservoir (Fuerteventura), built in 1943, which holds 350 000 m^3 of water, at present contains 300 000 m^3 of sediments (Torres Cabrera, 1995, unpublished data).

Therefore, there are three main causal factors of erosive activity in these islands: deforestation after the conquest (15th century), forest exploitation in the 19th century and the change from a socio-economic model mainly based on agriculture to another associated with the tertiary sector from the middle of the 20th century to the present day.

1.27.3 CURRENT EROSION PROCESSES

As can be observed in Figure 1.27.1, in the Canary Islands erosion due to rainfall is the type of accelerated erosion that most intensively contributes to soil loss, although wind erosion can be important in some islands.

1.27.3.1 Water Erosion

Approximately 43% of the surface area of the archipelago is affected by serious processes of accelerated water erosion involving some 323 000 ha. When describing serious accelerated erosive processes, reference is being made to those surface areas where more than 12 t ha^{-1} of soil are lost yearly due to rainfall (Table 1.27.1).

As observed, the islands of Fuerteventura and Gran Canaria are the most affected by water erosion processes, mainly owing to the sparse plant cover and extreme aridity and to intense overgrazing in the former case and to intense human pressure on the land from the beginning of the conquest in the case of Gran Canaria (Sanchez *et al*., 1995, 1998).

The islands least affected by these processes are La Palma and El Hierro, where several favourable natural factors (dense vegetation, high infiltration rate of soils, etc.) coincide with low human pressures on the land (Rodríguez Rodríguez *et al*., 1998a).

All the studies carried out (Rodríguez Rodríguez, 2001; Rodríguez Rodríguez *et al*., 2001a, 2002a,b, 2003; Guerra *et al*., 2003) show that in the islands, water erosion can take place via different mechanisms depending on the bioclimatic and edaphic characteristics.

In the rainiest areas on the windward sides of the largest islands, soils are andic, the plant cover is relatively dense and water erosion is continuous, associated with past changes in land use that have caused severe alterations in plant cover.

In the most arid leeward coasts and most eastern islands, with a predominance of carbonate and saline soils and sparse and sometimes no plant cover, erosive processes are more localized, coinciding with the intense southern rainstorms and also associated with changes in land use, in this case urban development and other tourist infrastructure, which often obstruct natural runoff drainage channels.

Over a period of 9 years, we have studied the production of sediments by sheet erosion in two areas representative of both climatic areas, using erosion plots. A summary of the results obtained is given below.

1.27.3.1.1 Windward Areas of the Highest Islands

The study was carried out on three erosion plots installed in January 1993 on Fulvic Andosols (Ultic Fulvudands). The plots were enclosed, allowing quantification of the production of solids and the volume of runoff. Each plot had an area of 200 m^2 (25 × 8 m), and all had collecting tanks equipped with HS flumes, capacity probes and ultrasonic devices connected to data collecting equipment.

TABLE 1.27.1 Surface area affected by severe water erosion processes (soil losses above 12 t ha^{-1} yr^{-1})[a]

	Surface area affected (km^2)	Proportion of total surface area (%)
Fuerteventura	987.0	59.4
Gran Canaria	885.2	56.7
La Gomera	174.2	47.1
Tenerife	853.2	41.9
Lanzarote	259.1	30.6
La Palma	56.6	8.0
El Hierro	15.8	5.9
Canary Islands	3231.1	43.4

[a]The data were obtained by remote sensing after extrapolating field measurements (from Rodríguez Rodríguez *et al*., 1998a, reproduced by permission of Geoforma Ediciones).

The mean rainfall on the plots during the years studied (January 1993 to September 2002) was around 625 mm yr^{-1} with a high monthly and interannual variability (Table 1.27.2). The most erosive rainstorms were concentrated in the months of February–March and October–November. The values of maximum intensity of the rain usually oscillated between 30 and 80 mm h^{-1}, with a high homogeneity in the mean intensity of the rainfall (7–15 mm h^{-1}).

The soil erodibility calculated by the method of Wischmeier *et al.* (1971) was very low ($K = 0.21 \pm 0.04$ t yr^{-1} MJ^{-1} mm^{-1}) for the soils of the plots.

On the first plot the soil was kept bare by removal of vegetation, with a slope of 24%, the second featured pine (*Pinus radiata* D. Don) with a slope of 13% and the third plot featured substitution vegetation (*Rubio periclymeni-Rubetum*) and revegetated laurel forest with a slope of 24%.

The runoff generated on these soils was relatively low. This led to little soil loss due to sheet erosion with a mean value of 9.0 t ha^{-1} yr^{-1} (bare soil), although a high interannual variability was also observed (Table 1.27.2). The greatest production of sediment was not clearly related to the most intense rain, but instead to the soil moisture content at the time of rain. The largest rates of sediment yield and runoff were observed when the rain fell on dry soil (Rodríguez Rodríguez *et al.*, 2002b).

Given the known hydrophobicity of the surface horizons of andosols, a high rate of runoff is generated when the raindrops fall on dry soil. This runoff carries the surface aggregates off the bare soil and is independent of the intensity of the rainfall.

Upon slow moistening of the soil, the high infiltration rate and the high water retention capacity of these soils lead to a very low degree of runoff and this is only generated when the volume of rainfall is high. Then it removes, by sheet flow, the moistened aggregates that have been fragmented by prior high intensity water-drop impact (Rodríguez Rodríguez *et al.*, 2002a).

1.27.3.1.2 Arid Leeward Areas and the Islands with Less Relief

In this case, the study was carried out on two erosion plots installed in January 1993 on Petric Calcisols (Typic Petrocalcids). The sizes of these plots were also 25 × 8 m (200 m^2) and the equipment used was similar to that described above.

The mean rainfall on the plots during the years studied (January 1993 to September 2002) is around 69 mm yr^{-1}, with a low interannual variability concentrated in highly localized spatial and temporal events from October to April (Table 1.27.3).

TABLE 1.27.2 Windward zones of the islands with the greatest relief: annual amounts of rainfall, erosion and runoff

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	Mean
Rainfall											
P (mm)	870	443	511	667	811	403	889	627	396	631	625
I_{max} (mm h^{-1})	60	31	242	84	45	52	82	43	43	14	242
Sediment yield ($t\,ha^{-1}\,yr^{-1}$)											
Bare soil	28.9	8.7	5.6	17.4	14.9	3.0	9.5	0.8	0.4	0.4	9.0
Natural vegetation	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Reforested pine	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Runoff (mm)											
Bare soil	94.4	80.9	156.3	155.9	105.6	77.4	70.4	5.5	5.0	3.3	75.4
Natural vegetation	5.5	1.3	0.7	2.3	1.4	0.8	1.2	0.6	0.4	0.6	1.5
Reforested pine	3.2	0.7	0.7	1.5	1.3	0.5	1.8	0.9	0.8	1.2	1.3

TABLE 1.27.3 Arid leeward zones and low-altitude islands: annual amounts of rainfall, erosion and runoff

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	Mean
Rainfall											
P (mm)	26.2	43.8	89.4	141.5	69.5	59.1	33.0	60.7	94.6	74.4	69
I_{max} (mm h^{-1})	33.6	31.2	57.6	36.6	24.7	28.8	31.2	50.4	31.2	5.4	58
Sediment yield ($t\,ha^{-1}\,yr^{-1}$)											
Bare soil	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.01	0.04
Natural vegetation	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.04	0.0	0.0	0.00
Runoff (mm)											
Bare soil	0.14	0.11	2.6	4.5	3.3	0.14	0.80	1.6	0.64	2.9	1.7
Natural vegetation	0.03	0.03	2.6	3.8	3.5	0.71	0.27	1.3	0.27	2.6	1.5

The maximum intensity of the rains in these conditions is low when the interannual average is taken into account. However, it must be borne in mind that in this type of rainfall regime, with sporadic rainstorms from low-pressure fronts, mean values do not reflect the erosive potential of the rains that occur with a high hourly intensity.

The soil erodibility calculated by the method of Wischmeier *et al.* (1971) is $K = 0.62 \pm 0.08 \text{t}\,\text{yr}^{-1}\,\text{MJ}^{-1}\,\text{mm}^{-1}$ for the soils of the plots.

In the first plot, the soil was kept bare by manual removal of vegetation, with a slope of 13%, and the second featured steppe vegetation of herbaceous ruderal communities, with a slope of 13 %.

Runoff always corresponds to less than 5% of the total rainfall and the generation of sediment is almost nil on both plots, regardless of the amount of rainfall and the intensity of the rain events, and is more related to the state of the soil surface and its antecedent moisture content (Table 1.27.3).

If, as mentioned previously, rains occur sporadically in this region but with a high intensity, in these conditions erosion is probably not a continuous process such as occurs for the Andosols in the northern areas. Instead, it is discontinuous with time, occurring in a pulse form with only quantitative importance in certain years or for some stormy episodes under specific circumstances of soil humidity and surface conditions.

This type of saline silty soil has a high capacity to form a crust up to 1.5 cm thick that seals the soil surface after the weak rains that usually precede the more intense rain events.

This sealing resulting from these rains produces a drastic reduction in infiltration during the most intense rains and an important amount of runoff is generated that gives rise to laminar flow on the sealing crust. This presents a mechanical resistance to rupture, does not break up and the solid particles do not separate. Therefore, most rain events produce a large amount of runoff but no erosion.

Only occasionally, in the case of consecutive intense rain events, are sediments generated by laminar erosion on plots, since the sealed crust loses its stability by moistening and individual particles break off and are transported by the laminar flow.

However, in most of these soils an intense erosive morphology can be observed in the form of rills and small gullies not representative of the sheet erosion seen on the plots. In these cases, as the length of the slope increases (slopes much longer than the lengths of the plots), its turbulence and speed also increase and the laminar flow is concentrated in streams which flow towards the drainage channels already present on the slopes or open new furrows and rills, since the concentrated flow achieves a sufficient shear stress to break the surface crust.

1.27.3.2 Wind Erosion

Approximately 35% of the surface area of the archipelago is affected by serious processes of wind erosion involving some 260 000 ha. When describing serious erosive processes, reference is being made to those

surface areas where more than 12 t ha^{-1} of soil are lost yearly due to wind (according to the wind erosion equation of Woodruff and Siddoway, 1965) (Table 1.27.4).

Wind erosion has the greatest incidence on eastern islands and leeward areas of higher altitude with frequent north-east and south winds, sandy soils with a high erodibility, a low content of organic matter and a weakly developed structure with small-sized aggregates highly susceptible to wind transport.

Northerly and north-easterly winds are most frequent not only in summer but also throughout the year with a mean frequency close to 55%. Moreover, since they have mean wind speeds of 6.3–7.0 ms^{-1} these can be considered as the components with the greatest erosive capacity.

In the areas of the island most susceptible to wind erosion, factor I (soil erodibility to wind) (Woodruff and Siddoway, 1965) oscillates between 0 t ha^{-1} (84% of non-erodible materials) for soils in relatively wet areas with a higher organic matter content (Cambisols, Vertisols) and 695 t ha^{-1} for sandy soils of the 'jables' (Arenosols) (Figure 1.27.1). However, two factors notably reduce the erodibility of soils in these regions, the existence of a surface crust on most of these and the presence of a stony surface layer in many regions. The presence of a surface crust and a stony surface layer helps to reduce wind erosion to very low levels in spite of the high erodibility of the soils and the high erosivity of the wind. The exception is found in sandy areas that do not have this surface crust or stony layer.

Agricultural activities produce important changes to the soil surface. Tillage breaks up the surface crust causing a pronounced destructuring of the soil, favouring its transportability by wind (Rodríguez Rodríguez *et al*., 2001b).

As can be observed in Table 1.27.4, most problems of wind erosion occur in the eastern isles, which are flatter and are more exposed to the action of dominant winds and where most soils have little plant cover and a high erodibility. Here, on ploughed soils, losses by wind erosion have reached 145 t ha^{-1} yr^{-1} (Rodríguez Rodríguez *et al*., 1998b).

1.27.4 SOIL CONSERVATION

The results of the assessment of accelerated erosion of soils from the Canary Islands indicate that different control measures are required in the two bioclimatic zones studied.

In the most northern, wetter areas, biological and agronomic measures are more appropriate to maintain a permanent plant cover on the soil, reducing the kinetic energy of the water drops, the main causal agent of the sheet erosion that predominates in this area, improving infiltration and replenishing the groundwater.

TABLE 1.27.4 Surface area affected by severe wind erosion processes (soil losses above 12 t ha^{-1} yr^{-1})[a]

	Surface area affected (km^2)	Proportion of total surface area (%)
Fuerteventura	1358	81
Lanzarote	441	49
Gran Canaria	444	29
Tenerife	366	18
La Palma	0	0
La Gomera	0	0
El Hierro	0	0
Canary Islands	2609	35

[a]The data were obtained by remote sensing after extrapolating field measurements, according to Woodruff and Siddoway (1965).

On the other hand, in the more arid areas of the island, the measures to control erosion should be mainly mechanical, aimed at shortening the slope length, preventing acceleration and concentration of the flow, the main agent that triggers the erosive rill and gully erosion processes dominant in these areas. Measures here are also aimed at channelling and exploiting the surface runoff generated.

Traditionally, two main types of measures have been used in the Canary Islands to fight erosion: afforestation and forestry–hydrological recovery of watersheds. The main objective of the afforestation has been not only to control water and wind erosion but also to improve the landscape, to help replenish the watershed and, ultimately, to control the overall desertification process.

At present, in addition to a government programme aimed at increasing plant cover in areas with a high risk of erosion and carrying out work to channel and correct water-courses, other actions are proposed which are aimed at fighting the causes of the problem. These are based on land management changes. The most important are the following:

- to encourage and support crop changes in agricultural areas at high risk of erosion by other crops which give more continuous cover during the year or with protective characteristics;
- to ban or to limit extensive cattle grazing on more fragile areas or those at greater risk of erosion;
- strict compliance with Environmental Impact Assessment legislation when building civil works ensuring that these do not trigger erosive processes;
- recovery and maintenance in working condition of traditional structures for soil and water conservation;
- careful planning of tourist and leisure activities.

1.27.5 CONCLUSIONS

Approximately 43% of the surface area of the archipelago (323 000 ha) is affected by soil losses above 12 t ha^{-1} yr^{-1} due to water erosion, and 35% (260 000 ha) is affected by serious processes of wind erosion.

The activities that have contributed most to the incidence of erosive processes on the islands are overgrazing and removal of plant cover, including deforestation, and mass felling of areas of forest that occurred in certain periods.

There are three main causal factors of erosive activity in these islands: deforestation after the conquest (15th century), forestry exploitations in the 19th century and the change from a socio-economic model mainly based on agriculture to another associated with the tertiary sector from the middle of the 20th century to the present day.

The present restructuring and control of extensive cattle rearing activities, reafforestation undertaken by government institutions, better control of forest fires and conservation of large areas of the islands have slowed erosion processes.

However, the constant economic growth on the islands and the often uncontrolled and disordered tourist development are resulting in large-scale occupation of the land by urban and transport infrastructures, producing drastic alterations in the water cycle by accelerating production of runoff and the incidence of water erosion.

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1.28

Portugal

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1.28.1 INTRODUCTION

In addition to its continental component on the western fringe of the Iberian Peninsula (37–42° N, 6–9.5° W), Portugal also includes the islands of the Azores archipelago (37–39.5° N, 25–31° W) and those of the Madeira archipelago (32.5–33° N, 16–17° W). Portugal occupies an area of about 92 000 km^2 and, with just over 10 million inhabitants in 2001, has a fairly high population density of about 110 inhabitants km^{-2}.

Over 70 % of the territory of continental Portugal lies at less than 400 m above sea level. There exists a strong contrast between the relief of the region to the north of the Tejo river, where the majority of mountain chains and plateaux are located, and the rolling landscape of the southern Alentejo and Algarve regions. Continental Portugal can be divided into three main geomorphologic units: the Hercynian Massif (*Maciço Antigo*), which has suffered repeated folding, granitization and metamorphism and has been greatly worn down by erosion; the *Orlas MesoCenozoicas*, which comprise a narrow belt of Secondary and Tertiary rocks (limestones, sandstones, clays) stretching along the coast from Porto to the Algarve; and the Tertiary basins of the Tejo and Sado rivers. The Madeira and Azores archipelagos are of volcanic origin and both have a pronounced relief.

The climate of continental Portugal ranges from Atlantic to Mediterranean but generally exhibits the pronounced hot and dry summer season (June to September) that is characteristic of Mediterranean-type climates. Rainfall typically has its origin in depression systems. Mean annual rainfall varies from 1000 to 2000 mm in the north-west and central mountain areas to less than 500 mm in the interior Douro valley and the Alentejo and Algarve regions. Inter-annual rainfall variability, however, is fairly pronounced. Although the rainfall intensities may assume very high values of up to 90 mm h^{-1} (Brandão *et al.*, 2001), the rainfall erosivity factor R is medium to high for about one-third of the continental territory (Figure 1.28.1).

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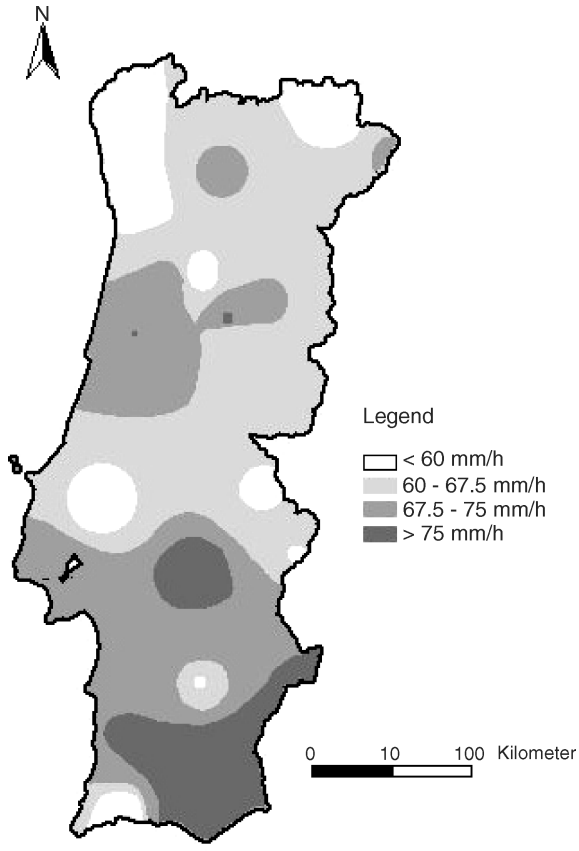

Figure 1.28.1 Rainfall erosivity (Pimenta *et al.*, 1997)

The prevailing soil types are Cambisols, Luvisols and Lithosols, occupying about 35, 25 and 15%, respectively, of the territory. Portuguese soils in general are regarded as of limited suitability for agricultural land uses. Following the national soil classification standards, about 80% of the soils have low cation-exchange capacity, about 70% have low organic matter content and about 80% have low pH.

Until the late 19th century, a large proportion of the rural area of continental Portugal was uncultivated or unproductive. Cultivated land occupied only about 20% of the country at the turn of the 20th century and now corresponds to about 45% (Figure 1.28.2). The Cereals Law from 1900 to 1920, the Wheat Campaign during the 1940s and the Agrarian Reform during the 1970s were important but rather unsuccessful attempts to increase cereal production, particularly in the Alentejo region, by augmenting the area of arable land and intensifying its use. The abandonment of the Agrarian Reform and the introduction of the Common Agricultural Policy (CAP) in the 1980s, plus widespread soil erosion problems and flooding of croplands, played a role in reducing the area under cultivation.

Since the end of the 19th century, the area occupied by forests and woodlands in continental Portugal has increased from approximately 7 to about 35% at the beginning of the 21st century (Figure 1.28.2). This increase involved all three prevailing forest types, i.e. maritime pine forests, eucalyptus plantations and evergreen oak woodlands (*montado)*. Eucalyptus species were introduced only in the mid-1900s but have, at an ever-increasing rate, become a landscape-dominating element at the expense of maritime pine forests in

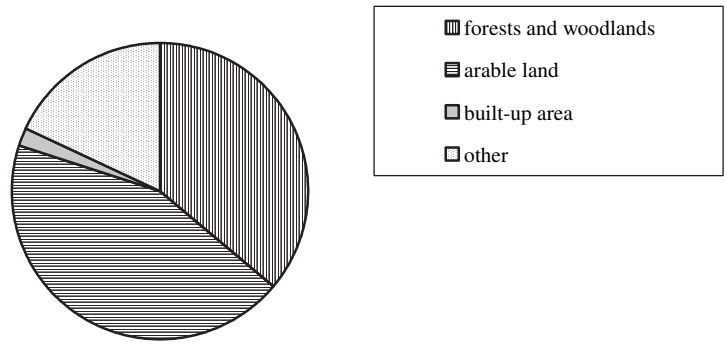

Figure 1.28.2 Main land-cover/use types in continental Portugal (DGA, 2000)

particular. Since the 1970s, frequent and sometimes very extensive fires have mainly 'consumed' maritime pine forests and played a key role in their replacement by eucalyptus forests.

1.28.2 HISTORICAL EVIDENCE FOR EROSION

Archaeological and sedimentological evidence at several Roman sites indicates the existence of periods of accelerated soil erosion during the Early Middle Ages. Also river channels suffered important aggradation, and wind erosion was responsible for the movement of sediments inland damaging medieval harbours and agricultural lands. In the 15th and 16th centuries, overgrazing and fire, deforestation for ship building and expansion of agricultural land were considered responsible for erosion of uplands and silting of the main rivers (Minho, Lima, Ave, Douro, Vouga, Mondego, Tejo, Sado, Guadiana), and advancement of coastal land into the sea (Mattoso, 1992).

1.28.3 SOIL EROSION ASSESSMENT

Given the natural conditions and climate of Portugal, water erosion in agricultural and forestry areas is a widespread phenomenon, aggravated by poor management practices (Figure 1.28.3).

Prompted by concern for the detrimental consequences of soil erosion on the productivity of agricultural lands, in the early 1950s the Portuguese government included in its Second Plan for Development a section on soil protection and conservation. By the end of that decade, Portugal started a soil conservation programme involving the installation of various standard Wischmeier plots. Owing to the initiative of Ernesto Baptista d'Araujo, an agronomist of the General Directorate of Hydraulics and Agricultural Engineering, the Vale Formoso Erosion Experimental Centre (VFEEC) was set up on a State-owned farm in the Mértola municipality (Alentejo). The main aim was to try to validate the universal soil loss equation (USLE) (Wischmeier and Smith, 1978) for soil management practices related to dry-farming cereal cultivation in Alentejo (Mértola) and vineyards in the Douro Wine region (Pinhão), and to promote soil conservation measures. Since the beginning of the 1960s, a number of Wischmeier plots have remained operational at the VFEEC, thereby providing an excellent database of about 40 years of continuous monitoring of various climate and soil loss and vegetation cover related parameters. This data set is amongst the most important and reliable of its kind in Europe (Cortez, 1987; Roxo, 1994).

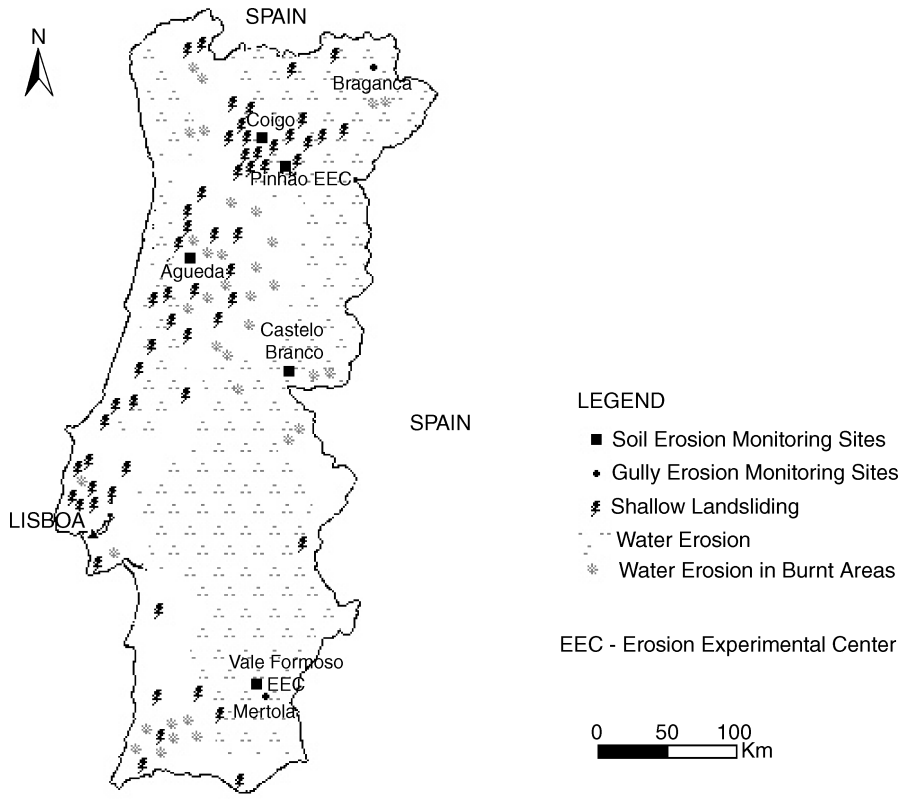

Figure 1.28.3 Soil erosion in Portugal

Results from the VFEEC, demonstrate very low soil losses in Alentejo (Table 1.28.1). The wheat and bare ground rotation registered values in the range 0.55–1.19 t ha^{-1}yr^{-1}, whereas on abandoned land under shrubland (cistus) the mean soil loss was 0.8 t ha^{-1} (Table 1.28.1) (Rosa, 1981; Roxo, 1994; Silva *et al.*, 1998). Results from the work of Silva *et al.* (1998) showed that over a period of 29 years only six rainfall events, out of a total of 475, were responsible for 64% of the total soil loss. Data from VFEEC may underestimated soil loss in Alentejo, since Vandaele *et al*. (1997) estimated gully erosion in the range 1.0–6.8 t ha^{-1}yr^{-1} in the Mértola region in soils with a wheat rotation (Table 1.28.1). Not until recently has research on soil degradation in Alentejo started to address the synergy of water erosion processes operating at different scales in the landscape explicitly discussing the role of land management practices like mechanised tillage in the development of Lithosols (Poesen *et al.*, 1994, 1996) and defining the geomorphological impact of land use change in the 20th century (Ferreira *et al.*, 2002).

Runoff and soil loss from five plots in vineyards was collected from 1979 to 1988, at Pinhão – Quinta de Santa Barbara, in Androsols, containing 60% by mass of rock fragments, in a vineyard, planted in 1971 in 45% gradient rows, up-and-down the slope. Annual average rainfall was 541 mm. Annual soil loss ranged from 0.005 to 1.9 t ha^{-1} (Table 1.28.1). The high rock fragment content of the topsoil might explain the low soil losses; also, 75% of the 10-year soil loss total was recorded in only four events, although 40% of all the events did not produce any runoff (Figueiredo and Ferreira, 1993; Figueiredo, 2001). Lopes *et al.* (2002) in Beira Baixa also reported very low soil loss on no-till plots either under pasture or lupines (0.019 t ha^{-1}), but higher volumes of runoff and soil loss were measured on plots under oats (Table 1.28.1).

TABLE 1.28.1 Summary of soil loss assessment on agricultural land

Location	Annual rainfall (mm)	Land use	Soil type	Slope angle (%)	Soil losses (t ha-1)	Measurement period	Plot size (m^2)	Reference
Water erosion								
Pinhão	540	Vineyard	Anthrosol	45	0.05–1.9	1979–1988	160	Figueiredo and Ferreira (1993) Figueiredo (2001)
Alto Douro	n.a.	Vineyard	Anthrosol	25	0.125	1994–1995	9	Oliveira (1995)
Castelo		Pasture	Cambisol	9	0.019	1991–1998	42	
Branco	740	Oats	Cambisol	9	0.1–0.2	1991–1998	42	Lopes *et al.* (2002)
	Control	Cambisol	9	1.6	1991–1998		42	
Vale		Wheat rotation	Lithosol	10–16	0.5–1.19	1961–1991	167	Silva *et al.* (1998)
Formoso	500	Shrubland (*Cistus*)	Lithosol	10–16	0.81	1988–1991	167	Roxo (1994)
EEC		Abandoned fields	Lithosol	10–16	0.18	1988–1991	167	Roxo (1994)
Terceira–Azores	1787	Pasture	Andosol	16	0.005–15	1996–98	1830[a]	Fontes *et al.* (2004)
Gully erosion							Surface area (ha)	
Bragança	740	Winter Cereals	Leptosol	15	6–28	1984–1995	n.a.	Vanderkerchove *et al.* (1998, 2000)
					1–5.2	1970	553	
Mértola	500	Wheat Rotation	Lithosol	10	1.4–6.8	1978	270	Vandaele *et al.* (1997)
					1.2–6.1	1985	270	

[a]Small experimental basin.

Fontes *et al.* (2004) assessed the hydrological behaviour of Androsols of the Terceira island of the Azores in two small drainage basins, under pasture for direct grazing in rotation with silage maize. The results show that erosion and sediment yield are very low under dense pasture pasture ($0.0005\ t\ ha^{-1}\ yr^{-1}$) but increase markedly when the soil is unprotected ($15\ t\ ha^{-1}$ for an 8-month period).

The results obtained in Castelo Branco and the Azores archipelago were also utilized for testing the USLE and OPUS models, respectively. The USLE, RUSLE and WEPP models were applied and tested by Silva and Silva (2001), Sebastião and Pereira (2002) and Fernandez *et al*. (2003), respectively, and applied to field-scale erosion on a rotating sprinkler-irrigation system.

As far as (agro-) forestry land uses are concerned, measurement of soil erosion by water has by and large been confined to the Serra do Caramulo in central Portugal. Erosion plots, repeat measurements of ground-level change and rainfall simulation experiments have been employed to assess the effects of wildfire, logging and ploughing and land-use change on soil loss and runoff (Coelho *et al.*, 1995, 1996; Walsh *et al.*, 1995; Shakesby *et al.*, 1996, 2002; Ferreira *et al.*, 2000). In eucalypt plantations, the total post-ploughing water erosion causes an estimated soil loss of $97–135\ t\ ha^{-1}$ after rip-ploughing (assuming a bulk density for removed soil of $1.0\ g\ cm^{-3}$) according to repeat ground-level change measurements, or ca $47\ t\ ha^{-1}$ according to erosion plot data (Coelho *et al.*, 1995; Shakesby *et al.* 2002). Wildfire-induced erosion prior to rip-ploughing amounts to an estimated $2–29\ t\ ha^{-1}$, based on new-burn pine and eucalypt erosion plot data, the range reflecting variability in soil depth and rainfall during different monitoring periods, together with differences in the amount of protective leaf and bark (Shakesby *et al.*, 1996). In total, therefore, the estimated amount of erosion following a wildfire and a downslope rip-ploughing cycle (including a tillage erosion component) is in the range $59–174\ t\ ha^{-1}$, if the basis for the calculations is accepted. In pine stands not subject to ploughing, the total estimated erosion following a wildfire cycle is up to about $29\ t\ ha^{-1}$ (Shakesby *et al.*, 2002).

Research seeking to quantify actual soil loss in Portugal has been rather fragmented in many respects and is generally of short duration and restricted to a few land uses.

Following the well-known CORINE-1992 assessment of potential soil erosion risk in Mediterranean Europe, which classified almost 70% of continental Portugal as presenting a high risk, similar approaches were developed and applied to produce various regional and nation-wide maps of potential soil erosion hazard (Coutinho and Tomás, 1994; Pimenta *et al*., 1997; INAG, 2003).

1.28.4 MAJOR ON- AND OFF-SITE PROBLEMS

For the majority of Portuguese soils, the major on-site consequence of runoff and soil erosion is general land degradation, involving reduced soil fertility, decreased infiltration and water storage capacity and diminished landscape value.

As is also evident from the CORINE-1992 assessment, soil erosion by water potentially is an almost nationwide problem in Portugal. The potential hazard due to the inherent climate, relief and soil characteristics is increased by profound changes in land use and land management practices that have occurred in the recent past. In the case of forests and woodlands in the central interior region in particular, the frequent occurrence of wildfires, sometimes affecting very large areas, exacerbates the soil erosion hazard.

Off-site effects of runoff and erosion include occasional losses of human life, damage to crops, silting of dams and reservoirs and other water bodies, eutrophication of various aquatic habitats, damage to public and private infrastructures and properties, loss of natural and cultural heritage values (sometimes leading to generalized landscape deterioration) and social and political disturbance (riots, clashes with local and national politicians).

Mass movements by landslides and mudflows, following long wet periods and intense rainfall, are responsible for widespread land degradation on the steep slopes of Portugal and also in the steep volcanic islands of Madeira and the Azores, accompanied by loss of lives and damage to properties (Figure 1.28.3).

1.28.5 SOIL CONSERVATION

Terracing is an important method of soil erosion control and water conservation, which decreases slope length and reduces damage by surface runoff. It has been used in the country for centuries. Thousands of small irrigated terraces fertilized with manure are common in the wetter and hillier areas of north-west Portugal. Since the 1960s, most of these areas have been abandoned and the majority of the terraces have collapsed, causing the rapid removal of soil by runoff.

In the Douro region, the soils and underlying schists have been greatly disturbed to form terraces since the 18th century, in order to support the first wine demarcated region of the world (*Região do Vinho do Porto*). This landscape of vineyard-covered terraces was recently assigned World Heritage Status. In the recent past, new vineyards are been planted in rows up and down the steep slopes but, owing to the high rock fragment content of the soils, very low soil loss and runoff are recorded.

Runoff and soil erosion are among the major environmental threats related to agricultural and forestry land uses in Portugal. Under the CAP reform, several policies have been designed and implemented in recent years, at European and national levels, to promote soil conservation and improve the agro-forestry sector.

Recent European policies and directives, such as the Water Framework Directive, the Nitrates Framework Directive and the European Commission Strategy for Soil Protection, and agro-environmental measures have addressed the issues of runoff and soil erosion and are being transferred to Portuguese legislation and starting to be implemented. The Ministry of Agriculture has issued a code of best practices for soil and water conservation in agriculture (DRDA/SRAP, 2001).

Portugal signed the United Nations Convention to Combat Desertification (UNCCC), after the Earth Summit of Rio de Janeiro in 1992. Under the National Action Programme to Combat Desertification (PANCD), the following strategic aims were adopted: soil and water conservation, rehabilitation of the most degraded areas and integration of desertification into national and sectorial policies for sustainable development.

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1.29

France

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1.29.1 INTRODUCTION

With an area of 550 000 km^2 on the western edge of Europe, France presents a large range of landscapes, climatic conditions, soils and land uses. Soil erosion was for a long time mainly considered as a problem of steep slopes and/or high-intensity rainfall, restricted to the Alps and the Pyrenees and to the Mediterranean area. No real attention was paid to soil erosion of agricultural soils on plateaux and in hilly areas before the end of the 1970s (Vogt, 1979). However, there is an increasing awareness of off-site impacts of runoff and soil erosion in regions occupied by intensive agriculture, even where slope gradients and rainfall intensities are relatively low (Auzet, 1987a; Papy and Douyer, 1991) and in particular in regions which are subject to extension of urbanization. A recent note, complementary to the 'Plan de Prévention des Risques d'Inondations' (Ministère de l'Écologie et du Développement Durable, 2003), emphasizes the risks related to the runoff generated in small 'peri-urban' catchments, typically <10 km^2, which now account for 75 % of the French communes. Among the risks, those related to mudflows and muddy floods at the foot of hillslopes or downstream of cultivated fields represent real disaster for inhabitants, ecosystems and water resources. More than 15 000 such events where registered during the period between 1985 and 2001 in France (Le Bissonnais *et al*., 2002b).

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In most of the areas which face severe off-site damage, risk prevention and soil and water protection are now considered major priorities by the government, the land and water authorities and the research institutes. Agricultural extension services, farmers and land managers also acknowledge the priority. Nevertheless, technical solutions are not sufficient: there is a real need for a better evaluation at catchment and small region scales, guidelines and coordination adapted to the large range of conditions without ignoring economic, environmental, political and social aspects.

At the national level, the Ministries of Agriculture and Environment completed three synthesizing surveys, in 1949, 1986 and 1996. The first is based on enquiries of State departmental services (Hénin and Gobillot, 1950), the second on a survey of published papers and unpublished reports (Auzet, 1987a,b) and the third on mudflow and muddy flood records and regional erosion risk modelling (Le Bissonnais *et al*., 1998, 2002a,b). In addition, two syntheses on water erosion in the southern French Alps have recently been published (Descroix and Gautier, 2002; Descroix and Mathys, 2003), and several studies are available at regional level, for example by Souadi *et al*. (2000) for Normandy, Le Bissonnais *et al*. (2004) for Picardy, Ballif (1999) for Champagne, Gascuel-Odoux and Heddadj (1999) for Brittany, Flota (1999), Auzet and Lemmel (2003), Auzet *et al*. (2005) and Heitz (2004) for Alsace.

After a brief introduction to some of the main aspects of geography, this chapter presents an overview of soil erosion in France: historical evidence, evolution since the end of World War II (which corresponds to major landscape changes due to both agriculture and urbanization) and the present-day situation. Key questions, information on available data, responses given by stakeholders and collaborative research in progress will be presented.

1.29.2 GEOGRAPHICAL CONTEXT

France has a large variety of landscapes, including high mountains in the east and south-west (Alps, Pyrenees) and extensive plains and plateaux of the Aquitaine and the Paris basins open to the west, with altitudes mostly below 200 m. Ancient Hercynian massifs lie around their rims. Among them, the Vosges and the Massif Central were thrown outwards at the time of uplifting of the Alpine range. The Massif Central contains many volcanoes. Massifs of the Tertiary Period (Jura, Pre-Alps) are made up of folded sedimentary rock, mainly limestone. Main river valleys mark the eastern part with the Rhone valley that opens towards the Mediterranean and, in the north, the western part of the upper Rhine floodplain.

Located between latitudes 41° and 52°, within the northern temperate zone, the climate is predominantly oceanic in the west and north, takes on continental characteristics in the north-east and is Mediterranean in the south-east. Mountain climate prevails in higher areas of the Alps and Pyrenees. Combinations of maritime influences, latitude, topography and altitude produce varied conditions.

These structures, and a long history of climate and land-use change, marked by migrations and regional variations of population density, land management, cultivation and practices (e.g. tillage), have resulted in a great variety of soils. To present a relevant typology relating to soil erosion and protection in a few pages of a book chapter is virtually impossible. Considering erosion, most sensitive soils are sandy and loamy loess derived soils – in all topographical locations – which cover considerable areas in France, but volcanic soils and marly regolith on steep slopes are also very sensitive.

Up to now, France has remained an agricultural country, accounting for more than 20% of the 15 EU Member States' total production, even though the agricultural sector employs only about 5% of the working population. Arable land covers 55% of the country. Wine and cereals are the two leading products, representing almost two-thirds of the agricultural production in 1999. Annual crops are predominant in 40% of agricultural areas (SCEES, 2001). The intensification of agriculture has occurred particularly in the last 30 years for economic reasons, with technological developments and the Common Agricultural Policy (CAP)

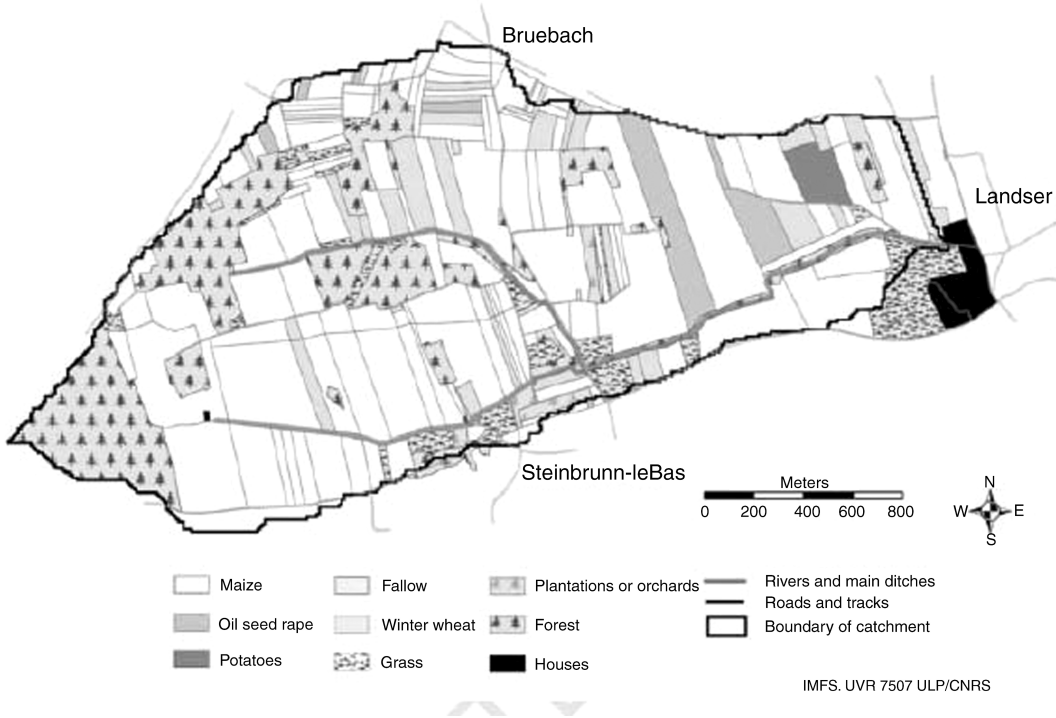

Figure 1.29.1 Land use in the Ibenbach catchment: Spring 2003 (Auzet and Lemmel, 2003)

of the European Union. The modifications in farming land are mainly due to the abandonment of the least profitable lands and to the extension of cities. On the income scale, winegrowers are at the top. Farmers in the Paris basin have the highest incomes among the field crop producers.

Even though the 1992 CAP reform was meant to contribute to overall grassland stabilization, at the level of regions of intensive agriculture, the decrease in pasture lands is still continuing (SCEES, 2001; Souchère *et al*., 2003). Moreover, in all the regions where large-scale farming industry has been developed, landscape changes do not consist only in an increase in field sizes and a decrease in pasture, forest and hedges: the trend to regional specialization in crop production, such as winter wheat in north-western France and maize in Alsace and the south-west, leads to a decrease in crop rotations, and finally a more homogeneous landscape. During the rainfall events, considerable areas of catchments are in the same state and can contribute to runoff and mudflows, as shown in Figure 1.29.1.

1.29.3 HISTORICAL EVIDENCE FOR EROSION

'From the end of the Middle Ages, inhabitants of alpine valleys have observed that erosion was linked to deforestation' (Descroix and Mathys, 2003). At the end of the 19th century, the main concern in France – mainly due to off-site damage and dam siltation – was in mountainous and/or Mediterranean areas. This led to a land restoration programme and reforestation which considerably diminished soil erosion in mountainous areas (Alps, Pyrenees and Massif Central), to the extent that it caused a debate on the consequences for

pastoral activities (Metailie, 1986; Cosandey and Muxart, 1989). The survey of Hénin and Gobillot (1950) was still mainly focused on erosion in mountains, but it indicated that some agricultural services of Departments had reported erosion problems in the valleys of the Loire, Yonne, Armançon, Seine, Aube, Marne, Meuse and lower Seine.

Evidence of soil profile truncation appears on detailed geomorphological and soil maps not only in mountain ranges but also on most of the plateaus and hillslopes. Interpretations of isotopic analysis of Quaternary sediments in valley bottoms of the Paris Basin point out important sedimentation since the end of the Gallo-Roman period. Some historical documents, such as 'Les cahiers de doléances' of the States General from the French Revolution (Vogt, 1953), indicate that soil erosion phenomena were not always restricted to the steepest slope or the highest intensity rainfall. In addition, the long tradition of transporting sediments from the bottom to the top of the hillslope in some vineyards shows that soil erosion is an endemic problem in France, which occurs in various forms. The main drivers are the interactions between climatic conditions and land use (Auzet *et al*., 1993).

1.29.4 CURRENT EROSION

As in other areas of Europe, there is an increasing awareness of erosion phenomena due to the off-site environmental impacts in several parts of France. Even though removal of soil is important and changes in topsoil characteristics are obvious in some landscapes, off-site effects are the most salient feature of erosion damage: mudflows and muddy floods originating from agricultural fields frequently enter urbanized areas that are located in adjacent valley bottoms or at the outlet of small catchments (Figure 1.29.2). On other hand, Water Authorities and Regional and Departmental Boards are aware of the impacts: sediment delivery to river courses and related degradation of water and ecosystems quality (Maret, 2004).

At the end of the 1950s, several papers reported a worsening of water erosion on agricultural soils in Picardy (Lefevre, 1958), in the 'Terrefort' near Toulouse (Brunet, 1957) and in the lower Rhone valley (Guennelon, 1956, 1958).

There is no doubt now that the data show evidence of a trend of aggravation of soil erosion-related problems in French regions:

- with concentration of annual crops or specialized crops such as vineyards and vegetables – despite the topography and the rainfall intensities that, however, remain important factors;
- where the extension of urban areas is greatest.

Since the law of July 1982, individual owners have to take out insurance against natural hazards. Through re-insurance mechanisms with a State guarantee, they can be reimbursed if the damage is related to a catastrophic natural hazard. In case of such damage, the mayor has to apply for the recognition of the state of catastrophic natural hazard and to fill out a form that allows the classification of the events. An inter-Ministry committee takes the decision, which is generally based on climatic data (rainfall event should exceed the decennial rainfall). Despite several uncertainties, the database CORINTE, now available on the Web, allows a very interesting study of location and cost (Figure 1.28.3; Le Bissonnais *et al*., 2002a,b). Nevertheless, there are now serious doubts as to the efficiency of this re-insurance mechanism and its effect on risk perception (Cartier, 1999, 2002; Heitz *et al*., 2004).

The density of mudflow and muddy flood records for France between 1985 and 2001 still appears very similar to the map roughly designed from information of various sources in 1986 (Auzet, 1987a, 1989). The

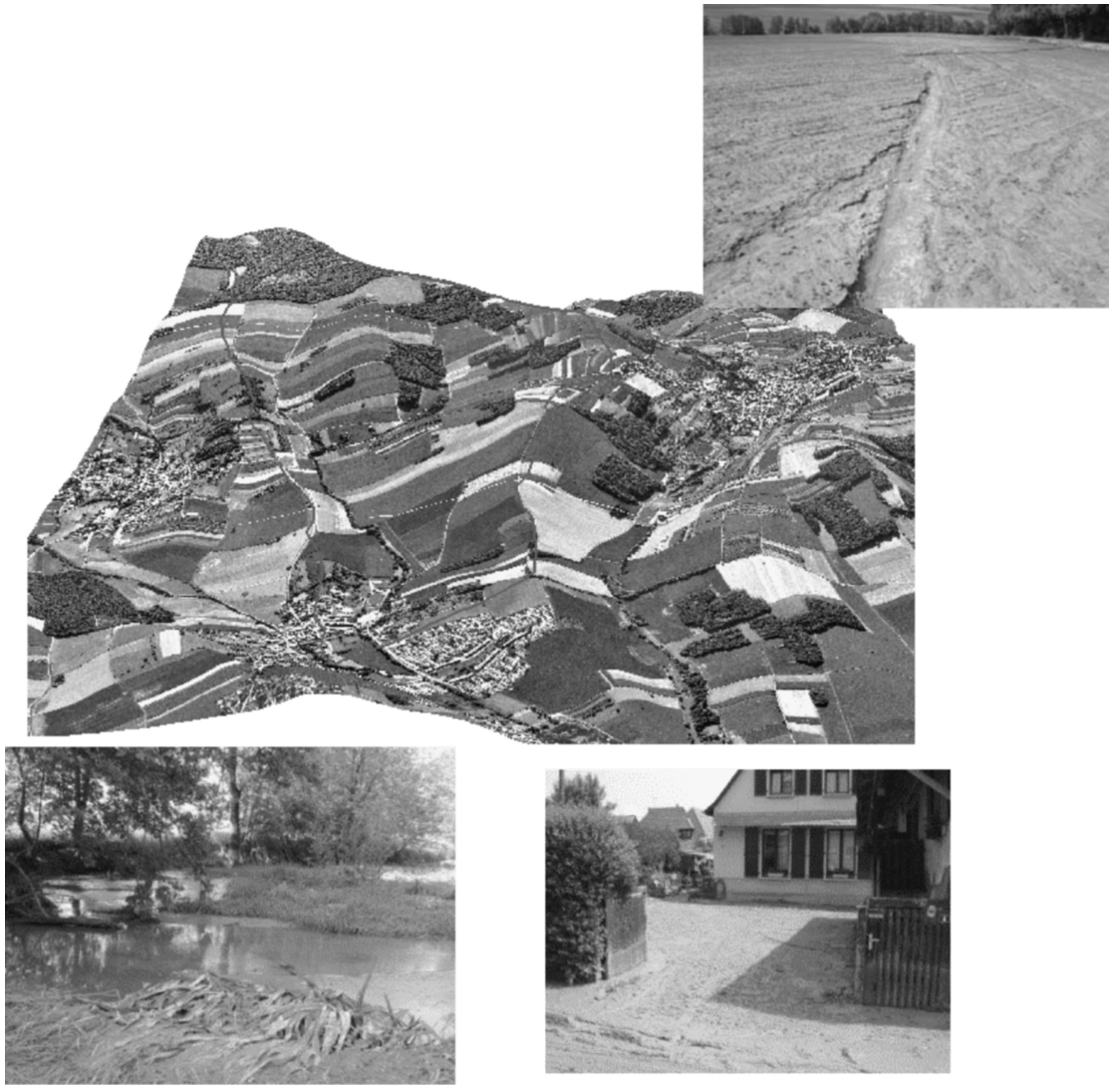

Figure 1.29.2 On- and off-site impact of soil erosion in a small catchment (Ibenbach catchment, May 2001)

Rhone valley and the Mediterranean area, the south-west part of France and the north-west of the Paris basin suffer from significant off-site erosion damage throughout the year. They are all regions where the increase in urbanization was the highest (Agreste Primeur, 2000).

The Mediterranean zone is particularly affected when high-intensity rainfall occurs during the autumn. In this area, rainfall erosivity is the highest (Pihan, 1988). Soil erosion problems are particularly noted after forest fires (Bonnet, 1984; Martin C *et al*., 1997) and in vineyards (Wainwright, 1996; Andrieux *et al*., 1998; Van Dijk, 2000). Steep slopes and non-cohesive materials such as marls, molasses and sandy soils are particularly subject to erosion. In noncultivated areas, vegetation cover, even sparse, is essential to prevent erosion (Clauzon and Vaudour, 1971; Cosandey and Muxart, 1987).

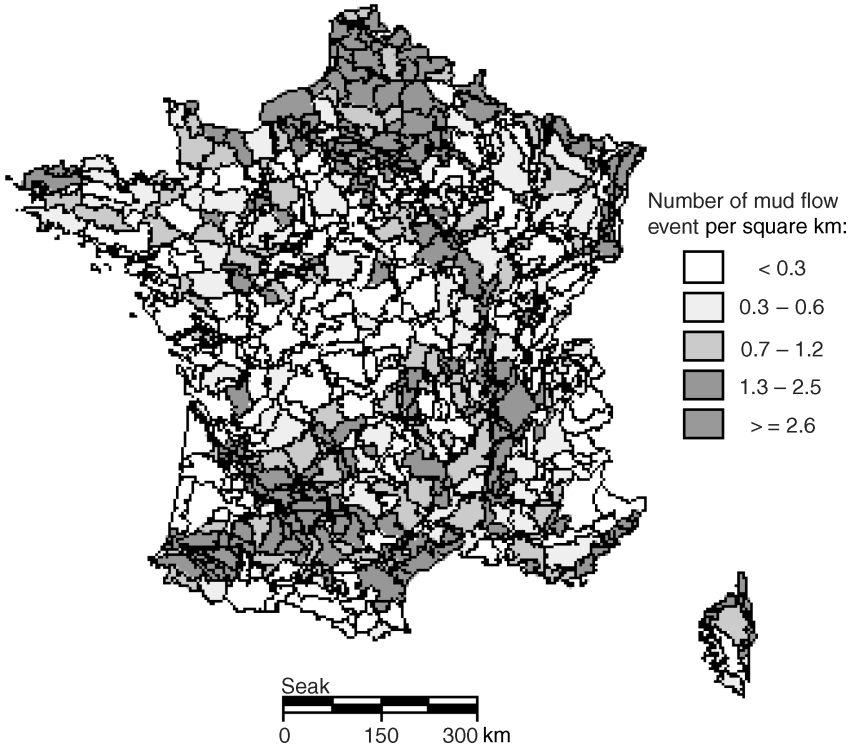

Figure 1.29.3 Mud flows and mud floods density in France between 1985 and 1995. (Reprinted from *Catena*, Vol. 46, Le Bissonnais Y, Montier C, Daroussin J, King D. Mapping erosion risk for cultivated soils in France, pp. 207–220. Copyright 2002, with permission from Elsevier)

In the south-west, the highest proportion of soil erosion damage occurs in spring and at the beginning of summer, in relation to low soil cover, high slope gradient and tillage operations (Brunet, 1957; Revel and Rouaud, 1985; Revel *et al*., 1990).

The north-western Paris basin appears as one of the most affected, despite low erosivity of rainfall and moderate slope gradient. The low structural stability of the loamy soils, the extension of the annual crops, the large amount of rain during the autumn and winter and the extension of urbanization are the main factors which account for the amount of damage (Auzet *et al*., 1990).

More limited parts with loess-derived soils are also strongly affected in Brittany (in autumn and in winter) (Gascuel-Odoux *et al*., 1996) and Alsace (during the spring) (Heitz, 2004).

The muddy flood records show that only a few regions are little affected, mainly in the central regions (Centre, Auvergne, Limousin) and the central part of the west (Poitou-Charentes). Grassland cover still remains important in these regions of extensive cattle and/or urbanization that has not considerably increased the vulnerability: erosion may occur, as on very sensitive volcanic soils, without leading to records of off-site damage.

In vineyards, there are considerable differences related to climatic conditions, slope gradients, soils, length of the parcels and farming practices (tillage, no tillage, grassed or not). Clearly, part of the Rhone valley and Mediterranean vineyards seem the most affected. Clear – and ambivalent – effects have been attributed to practices (Leonard and Andrieux, 1998; Ballif, 1999; Le Bissonnais *et al*., 2002b). In Alsace, the increased use

of a grass cover in inter-row areas has led to a decrease in soil erosion (Alajouanine, 2001). Research on the Roujan catchment, among others, clearly showed that agricultural practices may change the erosion considerably (Andrieux *et al*., 1998, 2001). Thunderstorms can lead to strong erosion if tillage was carried out a few days before. In all vineyards, the associated transfer of pollutants is mentioned. Nevertheless, there is a strong contrast of situations between regions, and moreover the runoff and erosion response appear ambivalent depending on the interactions between climate and practices and on the type of rainfall.

On other cultivated land, soil erosion appears to be related to different combinations of factors and a very frequent combination is heavy storms over bare soils. Of course, this occurs somewhat irregularly, owing to thunderstorm distribution. Erosion might be particularly high after tillage in recently sown spring crops (maize, sugar beet, vegetables) in Alsace (van Dijk *et al*., 2005), in northern France (CEMAGREF, 1986; Ouvry, 1986; Auzet *et al*., 1990) or in Brittany. These phenomena occur every year in several regions, but do not affect the same catchments each year. Another combination is frequent and widespread in the Paris basin in winter: catastrophic events may occur even with moderate rainfall intensities (Papy and Douyer, 1991; Le Bissonnais *et al*., 2002a; Delahaye, 2002). The combination of the amount of rain over the whole region during autumn and winter seasons, the predominance of loamy soils with low structural stability and the extension of annual crops that leads to limited cover in this period results in soil surface characteristics that strongly decrease the infiltration and the depressional storage. The high proportion of areas that may produce overland flow, even for low-intensity rainfall in agricultural catchments, leads to a high connectivity and rapid runoff concentration before reaching the permanent network (ditches or rivers), resulting in rill and ephemeral gully erosion (Auzet *et al*., 1995, 1998; Ludwig *et al*., 1995; Souchère *et al*., 1998, 2003; Cerdan *et al*., 2002; Blanchard *et al*., 1999). In all cases, the impact of agricultural practices is very important: soil surface characteristics can lead to very different runoff and erosion responses (Ouvry, 1986; Martin P *et al*., 1997, 2000, 2001).

In mountains, erosion has decreased since the end of the 19th century owing to restoration schemes and the decrease in agricultural land (Lilin, 1986). Economic activities no longer require as much space as they did, except now for transalpine roads and ski resorts (Descroix and Mathys, 2003). These areas are very sensitive to erosion owing to the interactions between weathering sensitivity (e.g. black marls, Descroix and Olivry, 2002; Maquaire *et al*., 2003), climatic factors such as freeze and intense rainfalls and demographic pressure. This leads to a strong contrast related to aspect. Landslides still remain an important risk (Maquaire, 2005). Owing to natural and artificial reforestation and torrent control, soil erosion decreased in the 20th century. However, reactivation by mountain urbanization emphasizes from time to time the constraints for land management (Descroix and Gautier, 2002). The complex interactions between slope processes that provide sediment sources and stream processes that are subject to torrential activity have led over recent decades to sedimentary deficits. The result was also an entrenchment of river beds that threatens infrastructures (bridges, embankments). This can represent severe problems in downstream areas, where alluvial deposits are abundant in more vulnerable areas (Brochot and Meunier, 1995; Brochot, 1998).

1.29.5 DEVELOPMENT OF KEY QUESTIONS

Until the end of 1970s, the main expertise on soil erosion and conservation was devoted to the RTM (Restauration des Terrains de Montagne) services, which are experienced in reforestation, and to the scientists working in tropical areas, mainly based at the ORSTOM (now IRD) research institute. According to the trends in international research, they developed methods based on experimental plots and rainfall simulation (Roose, 1977; Collinet and Valentin, 1979; Casenave and Valentin, 1989).

At the same time, the main concern of agronomic research was soil potential, soil fertility and soil physics – in order to promote agriculture and food production.

A lot of very interesting results were obtained with respect to the influence of soil surface characteristics on infiltration, structural stability and runoff generation at local and experimental plot scale. There is a clear need to save and reconsider the experimental results obtained. The question of the influence of soil surface characteristics on runoff generation and soil detachment remains a key issue, and also the relationships between agricultural practices and these soil surface characteristics (tillage, reduced or no tillage, soil biology and organic matter influences on infiltration and structural stability) (Martin P., 1999; Martin P. *et al*, 1998, 2001).

Despite the interesting work on extrapolation and scale issues, the key question of runoff transfer from fields to the foot of slopes and to the stream channel remains: none of the methods are satisfactory and it remains a matter of expert judgement. Nevertheless, flow on hillslopes cannot be considered as driven only by slope gradient. Particularly in the context of smooth slopes and relatively low-intensity rainfall, tillage direction appears to be a main driver (Ludwig *et al*., 1995; Souchère *et al*., 1998; Takken *et al*., 2001). In addition, many linear features can offer discontinuities or bypass runoff routes.

At the scale of small catchments, several questions have to be revisited. For instance, the question of tillage direction (or slope length) – which is commonly admitted as important – might be asked in a different way, when factors other than global slope gradient may considerably influence the flow direction and concentration. The question of the field size has probably to be reconsidered in the case of monoculture or dominant crop cover: in this case, limited field size may not lead to greater diversity of soil surface characteristics but to a higher number of headlands, tracks and dead furrows (Auzet and Lemmel, 2003). For similar reasons, the drainage that is generally seen as increasing the infiltration and limiting the overland flow has an ambivalent effect since it can lead to an increase in cultivated areas in catchments and a decrease in pasture.

Accordingly, key issues concern topsoil hydrology, topology of runoff routes and a better focus at the scale of small catchments to small regions.

1.29.6 AVAILABLE DATA

At the national level, databases for soil erosion modelling already exist, developed for the mapping of erosion risk at national or supra-national scales (King *et al*., 1998; Le Bissonnais *et al*., 1998, 2002a,b).

Available data for soil erosion studies, such as on relief, drainage network, climate, soils, land use and hydrology, already exist: it does not mean that they are all at the appropriate scale for modelling at catchment and small region scales, which are relevant when implementing operational measures, or that they are all easily accessible. Database development for that purpose remains a difficult task which is generally time consuming. Such databases were developed at least for the Nord Pas-de-Calais (IFEN *et al*., 1998), Haute-Normandie (Souadi *et al*., 2000), Alsace (van Dijk *et al*., 2005) and Aisne Department (Le Bissonnais *et al*., 2004). The current national programme for soil mapping called IGCS (Inventory, Gestion and Conservation of Soils) will provide soil databases at the scale of 1:25 000, which is relevant for modelling at regional scale. Together with the updating of Corine Land cover base and new climatic databases and digital elevation models, this should allow the improvement of the regional assessment of erosion risk in France.

No systematic efforts have been made at the national level to determine soil erosion rates. The attempts to evaluate rainfall erosivity or soil erodibility using USLE (Vogt and Vogt, 1979; Pihan, 1988) remained limited: the maps obtained clearly demonstrate the limitations of these approaches, particularly for loamy soils where complex interactions between climatic conditions, land use and practices result in soil surface degradation, runoff and erosion – even for low erosive rainfall (Auzet *et al*., 1990). More realistic are the maps produced by INRA (Le Bissonnais *et al*., 2002a) and by the EU-funded project PESERA (Gobin *et al*., 2004), although some questions remain for certain regions. In fact, these two maps, which show very similar erosion risk distribution in Europe, are based on very different modelling approaches but they use the same input databases.

Hence the low quality and resolution of input data may explain the apparent discrepancy observed between the model results and local expert knowledge.

Several groups have obtained experimental erosion data (see Chapter 2.4), but long series of measurements at catchment, field or plot scales are limited. Nevertheless, there are several catchments equipped in France for hydrological research or other purposes: the Draix (black marls) in Alpes de Haute Provence (Brochot and Meunier, 1995; Mathys *et al*., 1996), the Roujan (vineyard) in Languedoc-Roussillon (Voltz *et al*., 1994; Andrieux *et al*., 2001; Moussa *et al*., 2002), the Real Collobrier in the Var (Martin C, 1996; Martin C *et al*., 1997), the Mont Lozère in the Cevennes (Cosandey and Muxart, 1989; Bernard-Allée *et al*., 1991), several catchments in the Paris basin, in Haute-Normandie (Ouvry, 1986, 1992; Boiffin *et al*., 1988; Papy and Boiffin, 1988; Ludwig *et al*., 1996; Delahaye *et al*., 1999) and in Brie (Penven and Muxart, 1993). At plot scale, there are long series data for the vineyards of Champaigne (Ballif, 1999) and sandy loam soils of Cessières in Picardy (Wicherek, 2000).

It is difficult to compare data that come from plots of different sizes, with different methods of measurements – for that purpose, it is necessary to have more information than those generally given in papers. A very common range of high erosion rates at plot scale is between 2 and 3 t ha^{-1} yr^{-1} in the northern regions (Ballif, 1999; Wicherek, 2000; Cerdan *et al*., 2002) and up to several tens of t ha^{-1} yr^{-1} in the Mediterranean (Viguier, 1993; van Dijk S., 2000; Andrieux *et al*., 2001). At small catchment scales, several authors indicate erosion rates that can occasionally reach 20–50 t ha^{-1} yr^{-1} (Olivry, 1988; Ludwig *et al*., 1995; Penven *et al*., 1991; Cerdan, 2001; Cerdan *et al*., 2002; van Dijk *et al*., 2005).

1.29.7 MAJOR ON- AND OFF-SITE POLICIES TO COMBAT EROSION AND OFF-SITE PROBLEMS

The cost of the off-site impact is extremely high for France: data for the muddy flow records from 1985 to 1995 yield 5579 catastrophic events, with 3426 buildings affected, mainly individual houses (26 112) (Le Bissonnais *et al*., 2002b). Other impacts concern the agricultural fields themselves, the roads and other infrastructures, the river ecosystems, surface waters and in some areas the groundwater.

The increasing number of muddy flows and muddy floods since the end of the 1970s led the Ministries of the Environment and Agriculture to support surveys and action, which were set up in 1984 and 1985 in three small regions of the Paris basin (Pays-de-Caux, Artois and Ile de France) and one in the Midi-Pyrénées (Lauragais). They were initiated by local and departmental authorities. Depending on the context, the actions focused mainly on protection – basins and hydraulic measures – or on the development of actions and advice to promote better practices for farmers (Witkowski *et al*., 2004). These actions are generally supported by agricultural extension services, which in France are directly financed by farmers' taxes.

Although no specific legislation on soil protection exists in France, several instruments can be used for that purpose, among others:

- The PPR (Risk Prevention Plan) from 1995, which allow the creation of detailed descriptions of the country.
- The PLU (local town planning) describes the urban development plan of a city and defines land occupation and conditions for land use. This document also contains the permissions and the obligations concerning the road network and the 'natural' flow of water, and may plan the protection of trees and hedges.
- The SAGE (Water Management Scheme) introduced by law in 1992 defines the objectives in terms of water protection and aquatic ecosystems.
- Since 1995, the regrouping of land includes new plans for water and landscape protection.
- French water laws since 1992 divide the national territory into six large river catchments, for which action has to be taken to protect water bodies.

- The new law on natural risks (J.O., 2003) should lead to the constitution of departmental commissions of natural risks, which will have to delineate the areas of erosion risk where farmers and land owners will be obliged to apply soil protection measures against erosion.

They are many possibilities for territorial authorities to promote the prevention of soil erosion, particularly at regional and departmental level. In those regions where off-site impact of soil erosion is high, many actions have been initiated during recent years. For example, in Seine Maritime, where the frequency and intensity of on-site and off-site erosion damage are at the highest (Boardman *et al*., 1994), coordination of actions on a territory is a very important challenge. Generally supported by water authorities, new jobs have appeared during the last year to improve the organization of the complementary actions for best practices in agriculture and land management.

1.29.8 COORDINATION OF ACTIONS AND COLLABORATIVE RESEARCH

At present, the focus for water erosion in France is on cultivated areas. Implementing soil protection and mudflow prevention really requires adapted methods that do not exist at present, although the basic rules for soil protection against water erosion (such as surface coverage by residues or intermediate crops during the winter period) are well known. Effective methods, such as the reversion of vineyards or areas of annual crops to pasture or forest, are not considered for large areas. There is a real need for knowledge and for appropriate tools to evaluate scenarios and demonstrate their possible impacts.

The promotion of more favourable soil surface conditions on cultivated fields requires experimentation, evaluation and guidelines for agricultural practices: none probably is universally valuable for the entire country, owing to the very varied conditions of topography, soils, climate, land use and cultural habits. Advising on the best choice of crops and practices requires interdisciplinary competences. The question of the choice of crops and practices should not be restricted to the field level, but to the farm on one hand, and to the catchment on the other. To control erosion requires an ability to support the decision at both of these levels, by advice and incentives. However, even though fields are common units, farm territory and catchments are two distinct groups. The proposed measures should take into account this reality. Coordination of farmers' decisions at the catchment level has to be developed to take into account the neighbourhood effects of fields on overland flow and erosion (Papy and Souchère, 1993; Joannon *et al*., 2001; Mathieu and Joannon, 2003; Joannon, 2004).

Land management also requires a better understanding of runoff routes and associated sediment transport from fields to the catchment outlet. Field observations reveal the importance of the location of tracks and field access: often, owing to bypassing, the present situation allows for rapid flow transfers.

To support the evaluation and the decision, methods to survey rapidly the key points and to obtain the relevant spatial data will be very helpful, in addition to the further development of simple but realistic models to test scenarios and to allow the spatial representation of the output. Although these methods are being developed (Souchère *et al*., 2004), they are not easily available yet. Once solutions are found, the main stakeholders need to be convinced and actions need to be coordinated: clear input of sociological and economic arguments may help (Cartier, 1999).

Research on soil erosion has been developed by several groups in different institutes and universities. In the most affected regions, demonstration programmes already exist. Collaboration between end users, among them particularly agricultural extension services, and research groups was initiated. In addition, for several years, collaborative programme have existed at the national level. For cultivated soils, the three main research programs are RIDES (Ruissellement, Infiltration, Dynamique des Etats de Surface et Transfert des Sédiments) supported by the PNRH (National Research Program in Hydrology), GESSOL (related to the monitoring,

management and protection of soils and soil environmental functions) and RDT (Risk, Decision, Territory), both supported by the Ministry of Ecology and Sustainable Development. RIDES focus on the runoff/infiltration partition, with particular reference to the influence of local heterogeneities and their impact on the spatial distribution of sediment transport. GESSOL and RDT programme include socio-economic research. At the regional level, collaborations have been built with State and regional services as well as agricultural extension services. Research on torrential erosion and shallow landslides of the mountainous range is included in collaborative projects of the National Programme on natural and catastrophic hazard assessment.

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1.30

Belgium

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1.30.1 INTRODUCTION

Although a small country, Belgium has a wide variety of landscapes that is related to spatial variations in geology, soils, land use and topography. From a soil erosion perspective, three main regions can be distinguished (Figure 1.30.1a). The northern part consists of a large lowland plain (0–50 m above sea level) composed of Tertiary sands and clays, which are covered by late Quaternary aeolian sands. Isolated hills and sand-dunes occur locally. Soils are mainly sands and sandy loams. Slopes rarely exceed 2 %, with the majority less than 1 %. Land use is mixed although agriculture is more important in the west, whereas there are more forests in the east. Urban areas are also very important.

The central part of Belgium consists of a plateau with elevations ranging from 90 m on the northern edge to 200 m in the south near the Meuse River. The transition between the lowland plain in the north and the plateau in the centre is accentuated by a rise in topography, which is sharply expressed in the eastern part of the country but gradually levels out west of Brussels. The central plateau is also composed of Tertiary sediments. These are covered by Quaternary loess deposits up to several metres thick. The transition zone from the

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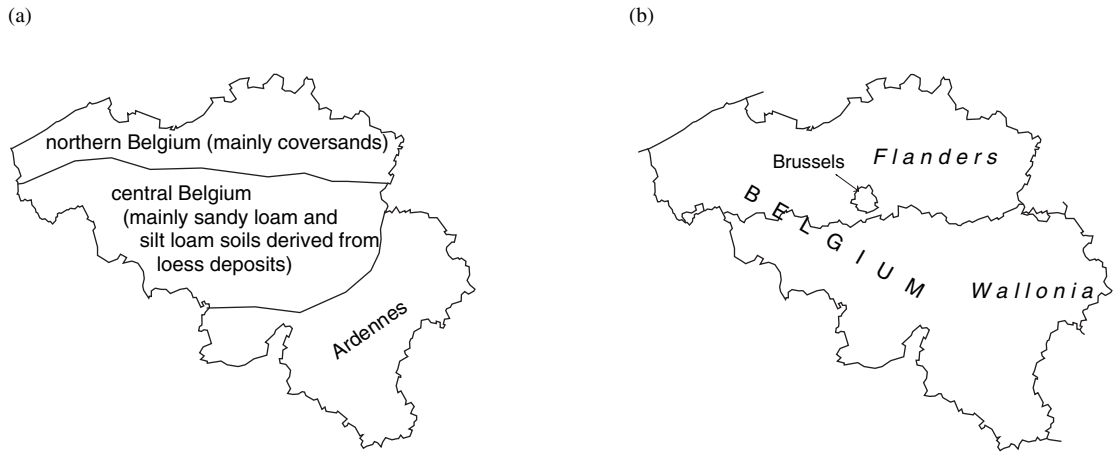

Figure 1.30.1 (a) Belgium subdivided into three regions with different erosional susceptibility (for explanation, see text). (b) The federal state of Belgium with three regions: Brussels Capital Region, Flanders Region and Walloon Region

northern plains to the central plateau is covered by sandy loam deposits. The central plateau is heavily dissected by south–north running rivers creating a hilly topography, especially in the eastern part of the country. In the west, the valley bottoms are wider and more isolated hills and hill ridges occur. Many slopes are steeper than 5 % and steepest slopes are in general 15–25 %. South of the Sambre and Meuse Rivers, the plateau of the Condroz region (200–300 m) is composed of south-west–north-east elongated sandstone hill ridges and limestone and shale dominated valleys. These Paleozoic formations are also covered by Quaternary loess deposits, less continuous compared with the area north of the Sambre and Meuse Rivers. The southernmost extent of these loess deposits borders the central part of the country to the south. Agriculture is the main land use in central Belgium, although large forests occur near Brussels and Leuven. The dominant crops are winter wheat, maize, sugar beet and potatoes.

Further to the south, the Paleozoic rocks of the Ardennes are characterized by thin Leptosols. They are mainly occupied by forest and pasture. Within this third region of Belgium, cropland is almost negligible. Slopes can be very steep (>30 %), especially near the incised river channels.

Mean annual precipitation in the northern and central part of Belgium varies between 700 and 900 mm, but in the Ardennes it can reach 1400 mm. Laurant and Bollinne (1976) estimated the mean annual rainfall erosivity for Ukkel (Brussels) at 649 MJ mm ha^{-1} yr^{-1} for the period 1934–73, a value that was updated by Verstraeten *et al.* (2001) to 677 MJ mm ha^{-1} yr^{-1} using detailed precipitation data for a 100-year period (1898–1997). Precipitation is well distributed throughout the year, yet rainfall erosivity shows a peak in July–August. About 70 % of the annual erosivity is concentrated in the period May–September (Figure 1.30.2).

In Belgium, the three regions (Flanders, Wallonia and Brussels) are responsible for environmental matters (including soil erosion, land and water management) (Figure 1.30.1b). Therefore, many of the discussions in this paper relate to either Flanders or Wallonia. Furthermore, the three federal regions do not correspond to the three physical regions discussed above.

1.30.2 HISTORIC EVIDENCE FOR SOIL EROSION IN BELGIUM

In many forests in central Belgium, large gullies can be observed as a relict of historic soil erosion (Arnould-De Bontridder and Paulis, 1966; Gullentops, 1992; Langohr and Sanders, 1985; Poesen *et al.*, 2000;

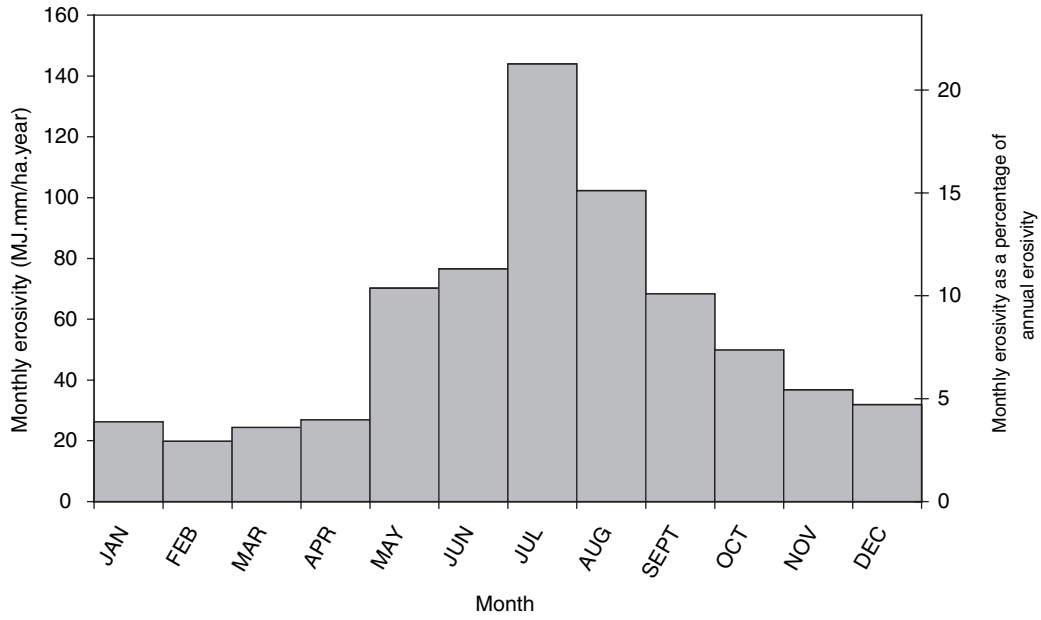

Figure 1.30.2 Distribution of monthly rain erosivity in Brussels (central Belgium). [Reproduced from Verstraeten G, Van Oost K, Van Rompaey A, Poesen J, Govers G, *Integraal Land- en Waterbeheer in Landelijke Gebieden met het Oog op het Beperken van Bodemverlies en Modderoverlast (Proefproject Gemeente Gingelom)*, 2001, by permission of Ministerie van de Vlaamse Gemeenschap]

Vanwalleghem *et al*., 2003). These gullies are mostly V-shaped and can be as deep as 4–10 m and more than 100 m long. If they become longer, they are often flat-bottomed. Different opinions exist as to when and why these gullies were formed. Some studies relate gully formation under forest to increased rainfall in the Atlantic period (e.g. Arnould-De Bontridder and Paulis, 1966; Bollinne, 1976), whereas others link them to the climatic anomaly of the Younger Dryas, before the forest was established (e.g. Langohr and Sanders, 1985). Poesen *et al*. (2000) and Vanwalleghem *et al*. (2003) conclude that, considering the spatial distribution of the gullies and their dimensions, and comparing these with present-day ephemeral gullies, the gullies under forest are probably formed by local disturbances in the forest during the Holocene, probably between the Iron Age and the Roman Period (Vanwalleghen *et al*. 2005; Vanwalleghen *et al*. 2006). Such local deforestations could have triggered significant runoff volumes that resulted in gully incision in downstream, still forested areas. Vanwalleghem *et al*. (2003) also report that many gullies were related to historic road patterns. Most of the typical sunken lanes in central Belgium could probably be related to this specific type of gully, a hypothesis already formulated by Gullentops (1992). Soil mapping and augering, e.g. done by the Belgian National Soil Survey, show that these gullies were not limited to present-day forested areas. Most of the concave dry valley bottoms under agriculture are characterized by thick colluvial deposits, often up to 5 or even 10 m thick in some cases (e.g. Vandaele, 1997; Bollinne, 1976; Gullentops, 1992; Nachtergaele *et al*., 2002). These valley deposits have been attributed to accelerated soil loss on hillslopes after the forest had been removed. This produced large quantities of sediment that was trapped within the valley-bottom gullies. Even nowadays, large gullies that are formed after extreme rain events will be filled with sediment originating from hillslope soil loss and tillage redistribution processes after a period of several decades (Nachtergaele *et al*., 2002). Within the forest, gullies are preserved because they were cut off from the sediment source after the local disturbance disappeared and the forest recovered.

Thick loamy alluvial deposits (up to 5 m) characterize many river valleys in central Belgium (Mullenders *et al*., 1966; Gullentops *et al*., 1966a,b; De Smedt, 1973). These alluvial deposits are a legacy of historic soil erosion since the introduction of cultivation in the area. This alluvium covers Atlantic peat (7800–5700 BP) and can mainly be attributed to the Subatlantic period (2600 BP–present), although it is already present in the Subboreal period (5700–2600 BP) in some places. For several rivers draining the Ardennes, the age and rate of recent alluvial deposits could be determined based on the presence of iron slag in the alluvium, which can be attributed to local iron smelting industries that were in operation since the 13th century (Macar, 1974). Since this period, alluvial deposits 1–2.6 m thick developed in the main river valleys, which corresponds to mean accretion rates of 1.4–3.7 mm yr^{-1} (mean 2.5 mm yr^{-1}).

Several studies have attempted to quantify historic soil erosion rates (Table 1.30.1). Various studies have concentrated on soil profile truncation measurements by soil augering to estimate historic soil loss rates in

TABLE 1.30.1 Historic evidence of soil loss and sediment export in central Belgium

Location	Source[a]	Temporal extent (years)	Spatial extent (ha)	Soil loss ($m^3\ ha^{-1}\ yr^{-1}$)[b]	Soil loss ($t\ ha^{-1}\ yr^{-1}$)[b]	Method[c]
Hammeveld 1 (Bertem)	(1)	1000	25	**6.0–8.0**	*8.1–10.8*	(a)
Hammeveld 2 (Bertem)	(1)	1000	25	**2.1–2.6**	*2.8–3.5*	(a)
Ganspoel (Huldenberg)	(1)	1000	110	**13.0**	*17.6*	(a)
Hammeveld 1 (Bertem)	(1)	1000	25	**3.0-4.0**	*4.1–5.4*	(b)
Hammeveld 2 (Bertem)	(1)	1000	25	**3.0–4.0**	*4.1–5.4*	(b)
Ganspoel (Huldenberg)	(1)	1000	110	**6.2**	*8.4*	(b)
Kinderveld (Bertem)	(1)	1000	210	**6.0**	*8.1*	(b)
Ganspoel (Huldenberg)	(2)	1000	90		16.9	(a)
Southwest of Leuven	(3)	1000	4000	**9.0**	*12.2*	(a)
La Hesbaye	(4)	2000	na	**3.25**	*4.4*	(b)
Momalle (Remicourt)	(5)	800	na	**2.2**[d]	*3.0*[d]	(c)
Gembloux	(5)	800	38	**6.4**[d]	*8.6*[d]	(c)
Gembloux	(6)	145–170	0.2–0.7	**8.5–10.4**	*12.8–15.6*	(d)
Bertem	(7)	1570	0.5		2.1	(d)
Bertem	(7)	49	0.5		5.5–9.8	(d)
				Sediment export ($m^3\ ha^{-1}\ yr^{-1}$)[b]	Sediment export ($t\ ha^{-1}\ yr^{-1}$)[b]	
Hammeveld 1 (Bertem)	(1)	1000	25	**3.3**	*4.5*	(a)
Hammeveld 2 (Bertem)	(1)	1000	25	**0.0**	*0.0*	(a)
Ganspoel (Huldenberg)	(1)	1000	110	**2.9**	*3.9*	(a)
Hammeveld 1 (Bertem)	(1)	1000	25	**0.0**	*0.0*	(b)
Hammeveld 2 (Bertem)	(1)	1000	25	**0.0**	*0.0*	(b)
Ganspoel (Huldenberg)	(1)	1000	110	**3.3**	*4.5*	(b)
Kinderveld (Bertem)	(1)	1000	210	**3.2**	*4.3*	(b)
Ganspoel (Huldenberg)	(2)	1000	90		3.9	(a)
La Hesbaye	(4)	2000	na	**1.75**	*2.4*	(b)

[a]*Sources*: (1) Vandaele, 1997; (2) Desmet, 1986; (3) Goossens, 1987; (4) Gullentops, 1992; (5) Macar, 1974; (6) Bollinne, 1977; (7) Gillijns *et al.*, 2005.

[b]Values in bold are original (measured) values; values in italic have been calculated assuming a dBD of 1.35 $t\ m^{-3}$.

[c]Methods: (a) soil augerings; (b) soil map; (c) colluvial deposits in dry valleys; (d) colluvial deposits in closed depressions; all four methods are based on volumetric measurements of historic soil erosion (profile truncation) and sediment deposits (colluvium).

[d]Underestimation of soil loss as only colluvial deposits have been measured, not what has been exported.

small catchments of central Belgium (Vandaele, 1997; Desmet, 1986; Goossens, 1987). Whenever volumetric erosion rates are reported, they were converted to erosion rates in $t\,ha^{-1}\,yr^{-1}$ using a mean dry bulk density of the topsoil of $1.35\,t\,m^{-3}$. Mean historic soil erosion rates vary between 2.8 and $17.6\,t\,ha^{-1}\,yr^{-1}$ for study areas ranging from 25 to 4000 ha, for a period of ca 1000 years. In some cases, data from the soil augering were also used to assess the long-term sediment export rates from small agricultural catchments. These sediment export rates vary between 0 and $4.5\,t\,ha^{-1}\,yr^{-1}$ for catchments ranging from 25 to 120 ha. A similar method has been used to reconstruct historic soil erosion rates on the basis of the Belgian soil map, which provides information about soil profile development and colluvial deposits, albeit in less detail (Vandaele, 1997). The results are comparable to the results from soil augering: soil loss rates vary from 4.1 to $8.4\,t\,ha^{-1}\,yr^{-1}$ for the past 1000 years, and sediment export ranges from 0 to $4.5\,t\,ha^{-1}\,yr^{-1}$ for the same period, and this for catchments ranging from 25 to 210 ha in size. Traditionally, it is assumed that soil erosion was only significant after the large forest clearings of the Middle Ages (11–13th century).

However, it can be questioned whether a time interval of 2000 years would not be more appropriate. This implies an important human impact on the environment from Roman times on. Indeed, the studies of historic gully erosion and floodplain sedimentation suggest that a cultivation period of 2000 years might be more appropriate for most of the loess areas in central Belgium. Gullentops (1992) also tried to quantify historic erosion and sedimentation volumes for the eastern part of the loess area in Belgium using soil profile data and the spatial extent of road gullies (sunken lanes). The eroded volume of the road gullies equalled $2000\,m^3\,ha^{-1}$, whereas erosion on hillslopes equalled $4000\,m^3\,ha^{-1}$. In total, $2500\,m^3\,ha^{-1}$ of colluvium was found, leaving $3500\,m^3\,ha^{-1}$that were probably exported to the rivers. Assuming a dry bulk density of $1.35\,t\,m^{-3}$ and a period of 2000 years, this corresponds to a mean erosion rate of $4.4\,t\,ha^{-1}\,yr^{-1}$ and a mean sediment export rate of $2.4\,t\,ha^{-1}\,yr^{-1}$. These values are in close agreement with the detailed studies by Vandaele (1997) and Desmet (1986) if we would recalculate these for a period of 2000 instead of 1000 years.

Macar (1974) reports on measured sediment volumes behind a bank that dammed small valleys. These volumes are attributed to historic soil erosion since the 12th century. The values range from 3 to $8.6\,t\,ha^{-1}\,yr^{-1}$. Finally, Bollinne (1977) and Gillijns *et al*. (2005) used colluvial sediment volumes in closed depressions to reconstruct past erosion rates. Within three such depressions, Bollinne (1977) estimated the erosion rates of the areas draining to the depression at 12.8–15.6 $t\,ha^{-1}\,yr^{-1}$ for the last 145–170 years. Gillijns *et al*. (2005) established a rate of $2.1\,t\,ha^{-1}\,yr^{-1}$ for the last 1570 years and 5.5–9.8 $t\,ha^{-1}\,yr^{-1}$ for the last 50 years.

1.30.3 CONTEMPORARY EROSION PROCESSES IN BELGIUM

Water and tillage erosion are the most important present-day erosion processes in Belgium. Especially in the central part of the country, which is characterized by loess deposits and hilly topography, both erosion processes are dominant (Figure 1.30.3). On the north-western and north-eastern sandy lowlands, wind erosion may occur. Soil loss due to crop harvesting is also mainly present in the central part of the country, although it is not limited to this region and occurs on every field where root and tuber crops are grown. Mass movements are of importance in the Flemish Ardennes (south of Ghent) and in the Pays de Herve (east of Liège).

1.30.3.1 Water Erosion Processes

Various types of water erosion processes operate in Belgium. These include splash, inter-rill, rill, ephemeral gully and bank gully erosion. In general, these processes are only of importance on agricultural land in central Belgium. In the northern part, the slopes are negligible and soils are much less erodible (sands and clays). Further to the south, in the Ardennes, land use is predominantly forest and pasture. Soil erosion in the forests

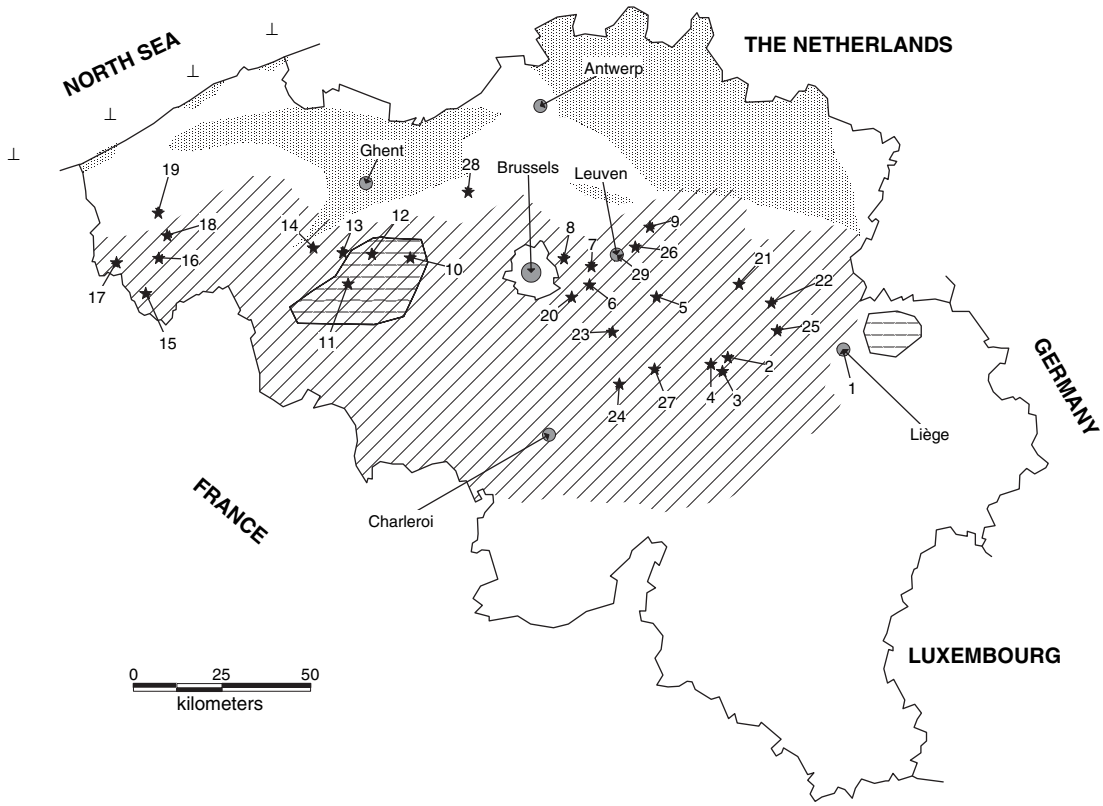

Figure 1.30.3 Areas prone to various soil erosion processes in Belgium. Oblique line shadings are areas prone to water erosion, dotted areas are prone to wind erosion and areas with horizontal line shading are prone to landsliding. Locations where erosion processes have been measured are indicated with a star: (1) Liège, (2) Braives, (3) Burdinne, (4) Hannut, (5) Hoegaarden, (6) Huldenberg, (7) Bertem, (8) Zaventem, (9) Holsbeek, (10) Herzele, (11) Maarkedal, (12) Zwalm, (13) Oudenaarde, (14) Kruishoutem, (15) Heuvelland, (16) Ieper, (17) Poperinge, (18) Langemark-Poelkappelle, (19) Houthulst, (20) Overijse, (21) Sint-Truiden, (22) Heers, (23) Chaumont-Gistoux, (24) Gembloux, (25) Remicourt, (26) Lubbeek, (27) Eghezée, (28) Dendermonde, (29) Leuven. see also Table 1.30.2

of the Ardennes is mostly limited to logging tracks, but no data on this exist. During recent decades, water erosion research has therefore essentially focused on the Belgian loess belt.

Table 1.30.2 provides an overview of measured water erosion rates in central Belgium. It should be noted that most of the measurements are volumetric and the results were converted to mass units using a single value of dry bulk density of the soil (dBD), which was often not measured. Not all authors report the original volumetric data, but for most of the studies these could be retrieved in original reports or when they provided a value for dBD. Many field measurements indicate that dBD of the topsoil varies between 1.1 and 1.6 t m^{-3}, with median values between 1.2 and 1.4 t m^{-3}. Govers (1986) measured dBD on 93 fields with a mean value of 1.38 t m^{-3}. On an experimental field in Huldenberg, dBD evolved after tilling operations from 0.95 to a constant value of 1.35 t m^{-3} (Govers and Poesen, 1986). To make all collected data comparable (but therefore not necessarily correct), we recalculated all the data assuming a constant dBD of 1.35 t m^{-3}. For that reason, for instance, the rill erosion rates from Bollinne (1977) of 14.8–26.5 t ha^{-1} yr^{-1} will be reported in Table 1.30.2 as 13.2–23.9 t ha^{-1} yr^{-1}.

TABLE 1.30.2 Current soil loss and sediment export rates for central Belgium

Location	Source[a]	Period	Temporal extent	Spatial extent (ha)	Soil loss ($m^3\ ha^{-1}\ yr^{-1}$)	Soil loss ($t\ ha^{-1}\ yr^{-1}$)[b]	Method[c]	Comments
Rill and inter-rill erosion								
Gembloux	(1)	1974–79	6 growing seasons	0.0089		**18.1**	(a)	Bare soil; 22.13 × 4 m; 6.5 % slope
Gembloux	(1)	1974–79	6 growing seasons	0.0089		**6.2**	(a)	Sugar beet; 22.13 × 4 m; 6.5 % slope
Gembloux	(1)	1974–78	5 growing seasons	0.0089		**4.0**	(a)	Winter wheat; 22.13 × 4 m; 6.5 % slope
Flemish Ardennes	(2)	1974	1 growing season			**10.3**	(a)	Sugar beet, 8.2 % slope, loamy sand
Flemish Ardennes	(2)	1974	1 growing season			**13.2**	(a)	Sugar beet, 14.6 % slope, sandy loam
Flemish Ardennes	(2)	1974	1 growing season			**0.8**	(a)	Sugar beet, 7.8 % slope, heavy sandy loam
Huldenberg	(3)	1983–84	1 year	0.75		**200.7**	(b)	Bare soil; 150 × 50 m; 13 % slope, max. 25 % slope
Rill erosion								
Poncia (Gembloux)	(4)	1969–70	1 winter period	0.2–6.0	**9.8–17.7**	*13.2–23.9*	(c)	4 winter wheat fields showing extreme rilling; 3–8 % slopes
Flemish Ardennes	(5)	1974	2 winter months	1.4	**9.3**	*12.5*	(c)	1 winter wheat field with severe rilling after an extremely wet period; 5 % slope
South-west of Leuven	(6)	1982–85	3 winter periods	na		**3.6**	(c)	86 winter wheat fields selected at random
Kinderveld (Bertem)	(7)	1997	1 event	250		**8.4**	(d)	Extreme event with return period of 10 years; mixed crop types
Heks (Heers)	(8)	1996	1 event	290		*35.2*	(d)	Extreme event with return period of 200 years; mixed crop types
Hammeveld 1 (Bertem)	(9)	1989–92	3 years	25	**3.3**	4.5	(d)	Two major events during measurement period; mixed crop types
Ganspoel (Huldenberg)	(9)	1989–92	3 years	110	**2.7**	3.6	(d)	Two major events during measurement period; mixed crop types
Kinderveld (Bertem)	(9)	1989–92	3 years	210	**1.0**	1.4	(d)	No extreme event during measurement period; mixed crop types

(*Continued*)

TABLE 1.30.2 (*Continued*)

Location	Source[a]	Period	Temporal extent	Spatial extent (ha)	Soil loss ($m^3\,ha^{-1}\,yr^{-1}$)	Soil loss ($t\,ha^{-1}\,yr^{-1}$)[b]	Method[c]	Comments
Ephemeral gully erosion								
Huldenberg	(3)	1983-84	1 year	0.75		**22.3**	(b)	Bare soil; 150 × 50 m; 13 % slope, max. 25 % slope
Kinderveld (Bertem)	(7)	1997	1 event	250		**6.8**	(d)	Extreme event with return period of 10 years; mixed crop types
Heks (Heers)	(8)	1996	1 event	290		*1.4*	(d)	Extreme event with return period of 200 years; mixed crop types
Hammeveld 1 (Bertem)	(9)	1989–92	3 years	25	**3.4**	4.6	(d)	Two major events during measurement period; mixed crop types
Ganspoel (Huldenberg)	(9)	1989–92	3 years	110	**3.2**	4.3	(d)	Two major events during measurement period; mixed crop types
Kinderveld (Bertem)	(9)	1989–92	3 years	210	**0.7**	0.9	(d)	No extreme event during measurement period; mixed crop types
Bertem	(10)	1995	1 event	218	**1.6**	2.2	(d)	After major rain event (return period 3 years) end of May
Bertem	(10)	1995	1 event	218	**1.9–3.6**	2.6–4.9	(e)	Same study area as previous; correction factor applied
Poperinge	(10)	1952–96	<6 months for 6 years	861	**0.9–1.6 (0.6–2.0i)**[d]	1.2–2.2 (0.8–2.7i)[d]	(e)	Analysis for six years (1952, 1959, 1973, 1985, 1988, 1996), randomly sampled; correction factor applied
Chaumont-Gistoux	(10)	1947–90	<6 months for 6 years	1095	**1.4–2.7 (0.6–4.5i)**[d]	1.9–3.6 (1.2–6.1i)[d]	(e)	Analysis for six years (1947, 1952, 1969, 1980, 1985, 1990), randomly sampled; correction factor applied
Sint-Truiden	(10)	1947–96	<6 months for 6 years	889	**2.3–4.6 (1.6–6.7i)**[d]	3.1–6.2 (2.2–9.0i)[d]	(e)	Analysis for six years (1947, 1957, 1975, 1983, 1989, 1996), randomly sampled; correction factor applied
Southwest of Leuven	(11)	1963–86	<6 months for 5 years	4000	**3.2–5.9 (1.2–10.2i)**[d]	4.2–7.6 (1.5–13.2i)[d]	(e)	Analysis for five years (1963, 1969, 1971, 1981, 1986), randomly sampled; no correction factor applied
Bank gully erosion								
Kinderveld (Bertem)	(12)	1988	10–20 years	100	**0.2–0.4**	0.3–0.6	(d)	Dry bulk density of 1.5 $t\,m^{-3}$ since most eroded material comes from more compacted subsoil

Ormendaal (Bertem)	(12)	1988	10–20 years	126	**0.25–0.5**	0.3–0.7	(d)	
Total measured water erosion rates in small catchments								
Hammeveld 1 (Bertem)	(13)	1989–92	3 years	25	**5.4 (1.6–11.9i)**[d]	8.1 (2.4–17.9i)[d]	(d)	Two major events during measurement period; mixed crop types
Hammeveld 2 (Bertem)	(13)	1989–92	3 years	25	**8.2(1.8–17.0i)**[d]	12.3 (2.7–25.5i)[d]	(d)	Two major events during measurement period; mixed crop types
Ganspoel (Huldenberg)	(9)	1989–92	3 years	110	**5.9 (0.2–10.7i)**[d]	8.9 (0.3–16.1i)[d]	(d)	Two major events during measurement period; mixed crop types
Kinderveld (Bertem)	(9)	1989–92	3 years	210	**1.7 (0.2–3.5i)**[d]	2.6 (0.3–5.2i)[d]	(d)	No extreme event during measurement period; mixed crop types
Huldenberg	(3)	1983–84	1 year	0.75		**223**	(b)	Bare soil; 150 × 50 m; 13 % slope, max. 25 % slope
Kinderveld (Bertem)	(7)	1997	1 event	250		16.7	(d)	Extreme event with return period of 10 years; mixed crop types
Kinderveld (Bertem)	(14)	1997–99	2 years	250		11.5	(d)	One extreme event during measurement period; mixed crop types
Ganspoel (Huldenberg)	(14)	1997–98	1.5 years	117		8.2	(d)	One major event during measurement period; mixed crop types
Heks (Heers)	(8)	1996	1 event	290		*40.3*	(d)	Extreme event with return period of 200 years; mixed crop types
Kinderveld (Bertem)	(9), (14)	1989–92 1997–99	5 years	226		*6.5*	(d)	Average values for two measurement periods, weighed according to measurement period length and catchment area
Ganspoel (Huldenberg)	(9), (14)	1989–92 1997–98	4.5 years	112		*8.6*		
Sediment export rates (sediment yield)								
Heks (Heers)	(8)	1996	1 event	290		*25.2*	(d)	Extreme event with return period of 200 years; mixed crop types
Kinderveld (Bertem)	(14)	1997	1 event	250		**8.0**	(f)	Extreme event with return period of 10 years; mixed crop types
Kinderveld (Bertem)	(14)	1996–99	3 years	250		**5.2**	(f)	
Ganspoel (Huldenberg)	(14)	1999–99	2.5 years	117		**1.5**	(f)	
Zwedebeek (Zwalm)	(15), (23)	1991–95	4 years	176		**1.7**	(f)	

TABLE 1.30.2 (*Continued*)

Location	Source[a]	Period	Temporal extent	Spatial extent (ha)	Soil loss ($m^3\ ha^{-1}\ yr^{-1}$)	Soil loss ($t\ ha^{-1}\ yr^{-1}$)[b]	Method[c]	Comments
Nederaalbeek (Maarkedal)	(15), (23)	1991–95	4 years	269		**1.7**	(f)	
Ter Erpenbeek (Herzele)	(15), (23)	1991–95	4 years	1172		**0.7**	(f)	
La Burdinale à Lamontzée (Burdinne)	(16)	1986	1 year	720		**0.7**	(f)	
La Burdinale à Marneffe (Burdinne)	(16)	1986	1 year	2600		**1.5**	(f)	
La Méhaigne (Eghezée)	(17)	1973–77	4 years	2044		**0.1**	(f)	Extreme dry period
Zwalm (Zwalm)	(18)			115000		**1.4**	(f)	Turbidity measurements (no samples)
Dijle (Bertem)	(19)	1985–86	1 year	73000		**0.7**	(f)	
Dijle (Leuven)	(20)	1959–60	1 year	74194		**0.3**	(f)	Dry period
Dender (Dendermonde)	(21)	1961	1 year	130000		**1.1**	(f)	
Dijle (Leuven)	(14)	1998–00	1.5 years	82000		**2.1**	(f)	Two wet winter periods
Dijle (Leuven)	(14), (19), (20)		3.5 years	77000		*0.9*	(f)	Average values for three measurement periods, weighed according to measurement period length
Maas (Luik)	(22)	1883–84	1 years	2018568		**0.1**	(f)	
Sterrebeek (Zaventem)	(23)	1996–99	3 years	7		**7.9**	(g)	
Nerm (Hoegaarden)	(23)	1996–99	3 years	20		**20.6**	(g)	One extreme event during observation period (return period >100 years)
Hammeveld 1 (Bertem)	(23)	1988–99	11 year	25		**5.9**	(g)	
Hannut	(23)	1996–99	3 years	73		**2.4**	(g)	
Ville-en-Hesbaye (Braives)	(23)	1996–99	3 years	103		**2.3**	(g)	
Overijse	(23)	1990–96	6 years	112		**11.6**	(g)	
Wolvengracht (Hoegaarden)	(23)	1996–99	3 years	148		**3.2**	(g)	
Ciplet (Braives)	(23)	1996–99	3 years	151		**2.0**	(g)	
Kleine Maalbeek (Zaventem)	(23)	1996–98	2 years	206		**3.5**	(g)	
Holsbeek	(23)	1988–99	11 years	226		**3.9**	(g)	
Ronebeek (Houthulst)	(23)	1993–95	2 years	783		**3.1**	(g)	
Bellewaerdebeek (Ieper)	(23)	1992–95	3 years	1050		**2.3**	(g)	

Munkbosbeek (Zwalm)	(23)	1993–97	4 years	1102	**1.2**	(g)	
Kemmelbeek (Ieper)	(23)	1936–82	66 years	1138	**2.6**	(g)	
Zouwbeek (Kruishoutem)	(23)	1994–96	2 years	1362	**1.5**	(g)	
Rooigembeek (Oudenaarde)	(23)	1994–96	2 years	1394	**2.3**	(g)	
Steenbeek (Houthulst)	(23)	1993–95	2 years	1908	**1.6**	(g)	
Broenbeek (Langemark-Poelkapelle)	(23)	1986–89	3 years	2423	**0.6**	(g)	
Douvebeek (Heuvelland)	(23)	1993–96	3 years	2651	**2.1**	(g)	
Vleterbeek (Poperinge)	(23)	1992–96	4 years	3235	**1.5**	(g)	
St-Jansbeek (Langemark-Poelkapelle)	(23)	1991–95	4 years	4873	**0.4**	(g)	
Huldenberg	(24)	1953–91	38 years	2.3	**5**	(h)	
Soil losses due to crop harvesting (see also Chapter 2.10)							
Belgium	(25)	1968–96	29 years	~95000	**8.7**	(i)	Sugar beet
Belgium	(25)	1990–96	7 years	na	**8.1**	(i)	Inuline chicory
Belgium	(25)	1996–97	1 year	na	**11.8**	(j)	Witloof chicory
Belgium	(26)	1999–2001	2 years		**2.1**	(i)	Potato
Belgium	(27), (28)	1995–96 2001–02	2 years	na	**15.8**	(i)	Carrot
Belgium	(28)	1995	1 year	na	**6.8**	(i)	Black salsify
Tillage erosion rates							
Kinderveld (Bertem)	(29)	1990	1 year	250	8.3	(k)	Catchment
Ganspoel (Huldenberg)	(29)	1990	1 year	110	9.3	(k)	Catchment
Kouberg (Hoegaarden)	(29)	1990	1 year	500	8.4	(k)	Catchment
Speelberg (Lubbeek)	(30)	1954–2000	46 years	2.4	4.0	(h)	Single field
Ganspoel (Huldenberg)	(30)	1954–92	38 years	1.5	10.3	(h)	Single field

[a] *Sources*: (1) Bollinne, 1982; (2) Pauwels *et al*., 1976; (3) Govers & Poesen, 1988; (4) Bollinne, 1977; (5) Gabriels *et al*., 1977; (6) Govers, 1991; (7) Steegen *et al*., 2000; (8) Takken *et al*., 1999; (9) Vandaele, 1997; (10) Nachtergaele and Poesen, 1999; (11) Vandaele *et al*., 1997; (12) Poesen *et al*., 1996; (13) Vandaele & Poesen, 1995; (14) Steegen, 2001; (15) Voet, 1997; (16) Lamalle *et al*., 1989; (17) Sine & Agneessens, 1978; (18) Huygens *et al*., 2000; (19) Huybrechts *et al*., 1989; (20) Gilles & Lorent, 1966; (21) Wirix & Lorent, 1966; (22) Spring & Prost, 1884; (23) Verstraeten & Poesen, 2001; Verstraeten, 2000; (24) Quine *et al*., 1994; (25) Poesen *et al*., 2001; (26) Biesmans, 2002; (27) Van Esch, 2003; (28) Soenens, 1997, (29) Van Oost *et al*., 2000a; (30) Van Oost, 2003.

[b] Values in bold are original (measured values); in the case of volumetric erosion assessments, values in italic have been recalculated from original values assuming a dry bulk density of $1.35\,t\,m^{-3}$ these values may therefore differ from the values reported in literature. Values are expressed per year for measurement periods longer than 1 year, otherwise these values are representative only for the measurement period. For rates of soil losses due to crop harvesting, the time period is one harvest.

[c] Measurement methods: (a) runoff plots; (b) experimental field; (c) field mapping – volumetric measurements; (d) field mapping at catchment scale – volumetric measurements; (e) aerial photographs – volumetric measurements; (f) suspended sediment sampling and continuous discharge measurements; (g) retention pond sedimentation rates; (h) ^{137}Cs meausurements; (i) based on factory data; (j) based on manually taken samples in the field; (k) tillage translocation model.

[d] i: values for individual years.

1.30.3.1.1 Inter-rill and Rill Erosion

These were the first water erosion processes that were intensively investigated in central Belgium. Bollinne (1982) measured splash detachment rates using splash cups and found a mean splash rate of 44 t ha^{-1} yr^{-1} for a classical rotation of sugar beet, winter wheat and winter barley. Poesen (1986) also measured splash detachment rates using splash cups. After correction for cup diameter (Poesen and Torri, 1988), these rates equal ca 70 t ha^{-1} yr^{-1} for a bare loamy topsoil and 175 t ha^{-1} yr^{-1} for a bare sandy topsoil. Splash distances are very short, however, and only a very small portion will contribute to net soil loss. On a bare experimental field site (0.75 ha), Govers and Poesen (1988) assessed the splash detachment rate at 166 t ha^{-1}, while only 3 t ha^{-1} were delivered to the rill system by splash.

Runoff and soil loss from Wischmeier plots on a 6 % slope were measured by Bollinne (1982) for three different treatments (bare soil, winter wheat and sugar beet) for 6 years (1974–79). Although mean annual soil loss values from these plots could not be calculated by Bollinne (1982) because of missing data, a minimal soil loss rate can be estimated by dividing the reported soil loss by the time period between the first and last measurement (5.5 years): 18.1 t ha^{-1} yr^{-1} for the bare soil, 6.2 t ha^{-1} yr^{-1} for the sugar beet and 4.0 t ha^{-1} yr^{-1} for the winter wheat. Maximum soil loss rates for one growing season are 77.8 t ha^{-1} for the bare soil (March–September 1974), 29.6 t ha^{-1} for sugar beet (May–September 1974) and 11.9 t ha^{-1} for winter wheat (November 1976–August 1977). A single extreme event in June 1974 caused a soil loss of 9.8 t ha^{-1} on the bare field plot and 11 t ha^{-1} on the field plot with sugar beet. During the growing season 1976, practically no soil loss was observed. On three other sites on the loess soils of the Flemish Ardennes (south of Ghent), Wischmeier plots were established in 1974 (Pauwels *et al*., 1976). All plots were cultivated with sugar beet and runoff and soil loss was measured during one growing season (June–October 1974). Total measured soil loss for this period ranged from 0.8 to 10.3 t ha^{-1} for fields having a slope of 7.8–8.2 %. Govers and Poesen (1988) measured soil losses from a bare experimental field site (0.75 ha) with a maximum slope of 25 % for 1 year (November 1983–October 1984). Total rill erosion was 152 t ha^{-1} yr^{-1}. If we also include the few observed gullies, the total concentrated flow erosion on this field plot was 174 t ha^{-1} yr^{-1}. Inter-rill erosion accounted for 22 % of the total soil loss from the field, which included direct splash into the rills (3 t ha^{-1}) and sheetwash (46 t ha^{-1}).

Rill erosion on cultivated fields has been monitored in several studies. Gabriels *et al*. (1977) estimated the total rill erosion rate at 12.5 t ha^{-1} on a winter wheat field (1.4 ha) in the Flemish Ardennes (south of Ghent) after a very wet winter period (213 mm from 20 October to 20 December 1974). Bollinne (1977) measured rill volumes on four winter wheat fields (0.22–6 ha) near Gembloux after the winter period 1969–70. Rill erosion rates varied between 13.2 and 23.9 t ha^{-1}. For the summer period, Takken *et al*. (1999) measured a total rill erosion rate of 35.2 t ha^{-1} following a rainfall event with a return period of >200 years in a 290-ha catchment, whereas Steegen *et al*. (2000) reports a rill erosion rate of 8.4 t ha^{-1} after a rainfall event with a return period of 10 years in a 250-ha catchment. All these measurements are not sufficient to reflect realistic mean figures of rill erosion in central Belgium. They only reflect rill erosion rates for fields or part of fields where rilling was present and/or only after rather extreme rainfall conditions. Govers (1991), however, surveyed 86 fields (winter wheat and bare soil) for three winter periods (1982–85), including fields where no or almost no rill erosion occurred. He found a mean rill erosion rate of 3.6 t ha^{-1} per winter period. Vandaele (1997) also surveyed rill erosion volumes in three agricultural catchments (25, 120 and 210 ha) for three years (1989–92) and found values of 4.5, 3.6 and 1.4 t ha^{-1} yr^{-1}. These values also include rill erosion for other crops (sugar beet, potato, maize, etc.), and rill erosion occurring in winter, spring and summer.

1.30.3.1.2 Ephemeral Gully Erosion

This has been studied extensively in the Belgian Loess Belt (e.g. Poesen *et al*., 1996; Vandaele, 1997; Nachtergaele, 2001). Most gullies are formed in zero- or first-order valleys where concentrated flow erosion is

intense enough to form channels that exceed the channel cross-section threshold of 929 cm^2 that is considered the lower limit to classify the eroded channels as gullies (Poesen, 1993). However, gullies can also form in linear landscape elements, such as plough furrows, parcel borders and access roads. With respect to the gullies in dry valleys, a clear distinction can be made between what is called 'winter gullies' and 'summer gullies' (Poesen and Govers, 1990; Nachtergaele *et al*., 2001). Ephemeral gullies formed at the end of the winter–early spring, are on average 53 cm wide and 26 cm deep (i.e. plough depth), whereas, ephemeral gullies that are formed after major rain events in late spring–summer tend to be more shallow (9 cm deep, i.e. thickness of the seedbed) but wide (307 cm) (Nachtergaele and Poesen, 2002) (Figure 1.30.4). The variations in gully depth and width can be explained by temporal changes in topsoil flow resistance. In some cases, gullies can be more than 2 m deep, namely when the much less resistant calcareous loess (i.e. soil parent material) is present at shallow depth. Summer gullies form with a single high-intensity rainfall event in a freshly tilled seedbed. Winter gully formation is a slow process as rainfall intensities are much lower and surface sealing is required to generate runoff that initiates gullies (Nachtergaele *et al*., 2001). The significance of ephemeral gully erosion for central Belgium has been illustrated by many studies. Using aerial photographs for different years, both Vandaele *et al*. (1997) and Nachtergaele and Poesen (1999) calculated the eroded volumes associated with ephemeral gully erosion at different locations in central Belgium. Their results indicate that ephemeral gully erosion can range from 1.2 to 7.6 $t\,ha^{-1}\,yr^{-1}$ (assuming a dry bulk density of 1.35 $t\,m^{-3}$) over a 6-month period (thus only winter or summer gullies). For individual years, the values range from 0.8 to 13.2 $t\,ha^{-1}\,yr^{-1}$, again for a maximum period of 6 months. It should be noted, however, that Nachtergaele and Poesen (1999) used a correction factor when applying the stereophotos, by comparing results with observed gully erosion volumes in

Figure 1.30.4 Severe muddy flooding in the streets of Hoegaarden after an extreme event (return period >100 years) on 29 May 1999. (Photograph copyright Gert Verstraeten)

the field. Using field surveys over a period of three years in three small catchments (25, 110 and 220 ha), Vandaele (1997) estimated the gully erosion rate at 4.6, 4.3 and 0.9 t ha^{-1} yr^{-1}, respectively. Takken *et al*. (1999) measured a total ephemeral gully erosion rate of only 1.4 t ha^{-1} following a one in 200-year rain event in a 290-ha catchment, whereas Steegen *et al*. (2000) reports a gully erosion rate of 6.8 t ha^{-1} after a rain event with a return period of 10 years in a 250-ha catchment. Finally, on the experimental field in Huldenberg, Govers and Poesen (1988) estimated the gully erosion rate at 22.3 t ha^{-1} yr^{-1}.

The ratio of gully erosion to total rill and inter-rill erosion varies greatly, depending on the temporal scale and the event size. For the small experimental field in Huldenberg, gully erosion accounts for only 10 % of total soil loss with an R/G ratio (soil loss by rill erosion to soil loss by gully erosion) of 6.8. However, this field was very small (0.75 ha) and with almost no flow convergence, so gullies could hardly develop. Data indicate that the relative contribution of gully erosion to total soil loss within a given catchment decreases with increasing return period of the rain event (Poesen *et al*., 2003). For an event in Heks with a 200-year return period (Takken *et al*., 1999), the R/G ratio was 25 and gully erosion was responsible for only 4 % of total soil loss. For rain events with return period of 1–10 years, gully erosion can be held responsible for 20–80 % of total soil losses. This also implies that gully erosion will be more important during winter periods when rain intensities are much lower than in summer periods. It is in these winter periods that gullies represent a very important linkage between upslope soil loss and downstream sediment delivery to rivers (e.g. Steegen *et al*., 2000). This could partly explain why observed sediment transport rates in rivers in Flanders are higher in winter periods, whereas the opposite is found for soil loss rates in small catchments (Steegen, 2001). Although ephemeral gullies are less important in summer, they still act as an important conveyor of sediment that may aggravate off-site damage (muddy floods).

1.30.3.1.3 Bank Gullies

Bank gullies in central Belgium can be observed near lynchets and terraces, but in particular along the banks of sunken lanes. They develop where concentrated overland flow from cropland crosses a weak spot in the bank (soil type, pipes, cracks). Although they can be spectacular and locally may represent an important erosion volume that contributes to downstream sediment delivery, few studies report on the intensity of this erosion process. Poesen *et al*. (1996) mapped and measured all the bank gullies in two small catchments near Leuven (100 and 126 ha). Mean bank gully volumes equalled 4 and 5.1 m^3 ha^{-1}, respectively. Assuming they formed over a period of 10–20 years and using a dry bulk density of 1.5 t m^{-3} (more compacted subsoil), this corresponds to mean annual soil losses of 0.3–0.7 t ha^{-1} yr^{-1}. This value is probably only representative for areas with many banks, such as the area between Brussels and Leuven. In many areas of the Loess Belt (eastern part of Flanders, Wallonia), however, land consolidation programmes have almost completely removed sunken lanes and lynchets, and bank gully erosion rates will be much lower.

1.30.3.1.4 Total Soil Loss by Water Erosion

Estimates are known for a number of catchments in the loess belt. In most cases, only rill and gully erosion have been measured (see above). Often, interrill erosion was estimated as a fraction of total soil loss. Mostly a value of 10 % of total rill and gully erosion was chosen (Steegen, 2001; Steegen *et al*., 2000; Takken *et al*., 1999). We applied the same contribution of inter-rill erosion to total soil erosion by water for the other studies. For the Kinderveld and Ganspoel catchments, two different monitoring periods exist. We therefore also calculated a mean value for these catchments using the data from the two periods, and by using the length of the period and the catchment area (which differs slightly between the two studies) as a weighting factor. For monitoring periods of several years, total water erosion rates (inter-rill, rill and gully) for the small agricultural

catchments in central Belgium (ca 100–300 ha) range from 6.5 to 12.3 t ha^{-1} yr^{-1}. For individual years, the range is much higher (0.3–25.5 t ha^{-1} yr^{-1}), whereas for individual events, the total soil loss by water erosion at a catchment scale may be as high as 20–40 t ha^{-1} yr^{-1}. The results for the Ganspoel and Kinderveld catchments clearly indicate that for monitoring water erosion processes, the length of the observation period is very important, owing to the importance of major rain events. In the case of the Kinderveld catchment, for instance, the event of 20 May 1997 (Steegen *et al*., 2000) was responsible for 56 % of the total erosion rate in a 5-year period. The low erosion rate reported for this catchment by Vandaele (1997), namely 2.6 t ha^{-1} yr^{-1} for a 3-year period, is due to the fact that no major rain event occurred during his observation period. This is in contrast to the three other catchments that he monitored during the same period.

For the Kinderveld catchment, we can also include the contribution of bank gully erosion to total soil loss, by assuming that the reported value of Poesen *et al*. (1996) is valid for the whole catchment as monitored by Steegen (2001) and Vandaele (1997). In that case, mean total soil loss by water erosion processes in Kinderveld equals ca 7 t ha^{-1} yr^{-1}.

1.30.3.1.5 Sediment Deposition

Sediment deposition by overland flow is an integral part of a catchment's sediment budget, yet it is often neglected in soil erosion studies. This is partly due to the fact that sediment deposition areas are limited in space. Steegen (2001) measured all sediment deposits within two agricultural catchments (117 and 250 ha) for 3 years and observed that only 2–3 % of the catchment is affected by important net sediment deposits. Owing to sediment deposition, only 10 and 55 % of the measured eroded volume was exported to the outlet of each of these two catchments, respectively (Steegen, 2001). Two broad types of sediment deposition can be identified in central Belgium: slope-controlled and typology-controlled sediment deposits, i.e. sediment deposits in furrows or in front of headlands, parcel borders and lynchets (Steegen, 2001). A specific type of typology controlled sediment deposit is sediment deposition in front of a vegetative barrier (Beuselinck *et al*., 2000), which can be of importance in the summer period when overland flow travels from a less vegetated field to a well vegetated field (e.g. wheat). On average, between 3 and 13 % of the eroded sediment in the two studied catchments (115 and 250 ha) was trapped in front of a parcel border, thereby indicating that the impact of parcel connectivity on sediment transfers is rather limited for these catchments (Steegen, 2001). In the past, with smaller fields and hence more field borders, the sediment trapping effect of field boundaries may have been as high as 15–20 %. The majority of the sediment deposits is slope controlled, however, and can be found at footslopes or in dry valley bottoms.

1.30.3.1.6 Sediment Export from Small Catchments

Sediment export from small catchments and sediment delivery to river channels are the result of water erosion processes and sediment deposition processes. A relatively large dataset on sediment export rates for a range of catchments in central Belgium exists, based on different methods (Verstraeten, 2000; Verstraeten and Poesen, 2001). For catchments ranging between 7 and almost 5000 ha, sediment yield ranges from 0.1 to 25.2 t ha^{-1} yr^{-1}. The minimum and maximum values are fairly extreme, however, as they were measured during an unusually long dry period and after a period with a very extreme event, respectively. A good correlation exists between catchment area (A in ha) and sediment yield (SY in t ha^{-1} yr^{-1}) (updated from Verstraeten and Poesen, 2001): $SY = 27A^{-0.42}$ with $r^2 = 0.51$ and $n = 29$ (excluding values from single extreme events). Verstraeten and Poesen (2001) discussed this variability in sediment yield and came to the conclusion that although catchment area explains more than half of the observed variability, it was not a valid parameter to link with sediment yield. Catchment geomorphology and the within-catchment spatial distribution of slope steepness, soil erodibility and land use were much more important.

1.30.3.2 Tillage Erosion and Soil Redistribution

The relevance of tillage operations for redistributing soil within the landscape has been recognized in many studies worldwide. Tillage erosion is especially important in the hilly regions of central Belgium. Several tillage experiments have been set-up in the loess area near Huldenberg to study soil displacement rates by various tillage operations (e.g. Govers *et al*., 1994; Van Muysen and Govers, 2002; Van Muysen *et al*., 2002; Van Oost *et al*., 2000b). These studies have resulted in the assessment of current tillage transport coefficients (k_{till}) for different tillage implements. Van Muysen and Govers (2002) found a mean k_{till} value of 123 kg m^{-1} for a rotary harrow and seeding equipment. For a mouldboard plough with a plough depth of 30 cm, Govers *et al*. (1994) found a k_{till} value of 234 kg m^{-1}, whereas Van Muysen *et al*. (2002) obtained 254 kg m^{-1}. Van Muysen *et al*. (2002), however, noted that in the case of reduced tillage depth, k_{till} values were much lower (e.g. only 50 kg m^{-1} for a mouldboard plough with a tillage depth of 15 instead of 30 cm). Tillage depth increased following World War II owing to the use of more powerful tractors (Van Oost *et al*., 2000a). This, and other changes in tillage intensity, imply that since the 1950s, mean annual k_{till} values have risen from ca 500 to 700 kg m^{-1}, assuming an average of one mouldboard plough operation, 1.5 chisel plough operation and one harrow operation each year. These values are in close agreement with mean annual k_{till} values that were derived from ^{137}Cs measurements and a ^{137}Cs conversion model on various fields in central Belgium (Van Oost *et al*., 2003). For fields where the introduction of heavy machinery took place much later, for example the Speelberg site near Leuven in the late 1970s, ^{137}Cs derived long-term k_{till} values (50 years) are lower (ca 350 kg m^{-1}; Van Oost, 2003).

Tillage erosion, therefore, has become important with the introduction of mechanization in agriculture. This can also be concluded if spatial patterns of tillage erosion and water erosion are compared with observed spatial patterns of ^{137}Cs and soil profile development. The spatial patterns of ^{137}Cs on complex landscapes in central Belgium clearly indicate that for the last 40–50 years, tillage operations have been the dominant processes changing the landscape (e.g. Van Muysen, 2001). Vandaele *et al*. (1996) came to the same conclusion by comparing sequential photographs from 1947 and 1991 for a field in central Belgium. However, if one looks at the spatial distribution of soil profile truncation and colluvial deposits, it is clear that in the long term (1000 years or more), water erosion processes are the key processes for shaping the landscape (Van Muysen, 2001). The impact of tillage in modifying the landscape is very clear near parcel borders, where it may create a bank. Many of these linear features that can nowadays be observed in central Belgium have been formed by annual tillage operations. Near parcel borders, tillage erosion rates can often exceed 50 or even 100 t ha^{-1} yr^{-1} (Van Oost, 2003). On slope convexities, tillage erosion rates can easily be higher than 10 or even 20 t ha^{-1} yr^{-1}, whereas in concavities, tillage deposition rates can be of the same order of magnitude (Van Oost, 2003). For two fields north of Leuven and in Huldenberg, the tillage erosion rate was estimated at 4.0 and 10.3t ha^{-1} yr^{-1}, respectively, for the parts of the field subject to soil loss (Van Oost, 2003; Van Oost *et al*., 2003). At the scale of the field, no net soil loss takes place due to tillage. Mean tillage erosion rates are therefore averaged for the part of the field where there is net erosion (Table 1.30.2). Doing so, Van Oost *et al*. (2000a) calculated mean annual tillage erosion rates of 9.3, 8.3 and 8.4 t ha^{-1} yr^{-1} for catchments of 100–500 ha with complex topographies.

1.30.3.3 Soil Losses Due to Crop Harvesting

Recently, soil loss due to crop harvesting (SLCH) was assessed for Belgium (Poesen *et al*., 2001) (see also Chapter 2.10). Mean SLCH values per harvest equal 8.7 t ha^{-1} for sugar beet, 2.1 t ha^{-1} for potatoes, 8.1 t ha^{-1} for inuline chicory, 11.8 t ha^{-1} for witlof chicory, 15.8 t ha^{-1} for carrots and 6.8 t ha^{-1} for black salsify. These are average values and SLCH can vary considerably, in both space and time. For the period 1968–96, mean annual SLCH for sugar beet ranged from 4.4 to 19.5 t ha^{-1} per harvest, whereas for the same period the value

for individual fields ranged from 1 to 100 t ha^{-1} per harvest. For the period 1991–2000, the annual average area of crops leading to SLCH (sugar beet, chicory, potatoes, carrots) in Belgium amounted to 175 000 ha. Sugar beet and potatoes are, with 95 000 and 57 000 ha, respectively, the most important crops. The total area of root and tuber crops corresponds to 23 % of the total Belgian cropland. Thus, on average once every 4 years, a root or tuber crop is being cultivated on every field. Using the mean SLCH figures for every crop and the mean annual areas on which they are grown, the mean annual soil loss by SLCH on cropland in Belgium can be assessed at 1.4 t ha^{-1} yr^{-1}. However, for the areas in central Belgium, which are already prone to high soil loss rates due to water erosion and to high soil redistribution rates by tillage operations, at least 31 % of the cropland is used for roots and tubers. In this region, almost once every 3 years a root or tuber crop is sown and the mean annual soil loss rate by SLCH for central Belgium can be estimated at 1.8 t ha^{-1} yr^{-1}. On certain fields, however, a root and/or tuber crop will be sown more frequently, sometimes even once every 2 years. In such cases, mean annual SLCH can be as high as 3–5 t ha^{-1} yr^{-1}. For the areas where soil erosion by water or tillage is negligible owing to the lower relief (e.g. northern Flanders), SLCH will be the most important soil erosion process.

1.30.3.4 Mass Movements

Although Belgium has, in general, gentle slopes and does not appear as an area susceptible to landsliding, mass movement features can be observed (Figure 1.30.3). In the Flemish Ardennes, south of Ghent, at least 135 old, deep-seated landslides (shear surface >3 m deep) are observed (e.g. Vanmaercke-Gottigny, 1980; Ost *et al*., 2003 Van Den Eeckhant *et al*. 2005). Rotational slides are the dominant landslide type in the region. In some cases these rotational slides develop downslope into flow-type movements. Some of the landslides are still active or have been reactivated. A survey by Ost *et al*. (2003) shows that the mean landslide area is 5 ha, with a mean scarp width of 250 m and an average length of 200 m. These landslides can be related to the geology of the area, which consists of alternating Tertiary deposits of clay and sand. For the majority of these landslides, the sliding plane lies in the clay layers that are rich in smectites. The slope angle also is an important factor explaining the presence of landslides, with the majority of slides having a pre-landslide slope of around 15 %. Important rain events are considered as possible triggering factors. Owing to the landsliding activity, many roads and even houses are seriously damaged. In addition to these large landslides, smaller and shallower movements affect the region.

In the eastern part of the country, i.e. the Pays de Herve east of Liège, several large landslides were observed (Demoulin *et al*., 2003). With an average landslide area of 6.4 ha and a pre-landslide slope of 12 %, it seems that their morphological and topographical characteristics are similar to those of the landslides found in the Flemish Ardennes. Another similarity is the important role of geology, i.e. Cretaceous deposits consisting of clay and sand. Radiocarbon dating revealed that some of the landslides were initiated around 2000 BP. Nowadays most of them are inactive.

1.30.3.5 River Bank Erosion

This process is not so important for most of the rivers in Belgium as many rivers, particularly the larger ones, are diked and have stabilized banks. Because of that, few studies have been made on this subject. One of the only rivers draining central and northern Belgium that has still a more or less ‘natural’ structure without bank protection or embankments is the River Dijle. Vandaele *et al*. (2002) measured the evolution of the river meanders and banks for the period 1970–90 using aerial photographs. They concluded that minimal river bank erosion rates vary between 0.02 and 0.06 m^3 m^{-1} yr^{-1} along the entire river length, which corresponds to a mean retreat rate of 0.1 m yr^{-1} for the active sections. This process, however, does not result in the widening of the river channel or in a lateral retreat of the banks, but rather in a downstream movement

of river meanders at an annual rate of 0.2–1 m. The downstream displacement rate of river meanders has also been measured for three smaller rivers in Flanders, with values ranging from 0.2 to 0.3 m yr^{-1} (Van Liefferinge *et al*., 2002).

1.30.3.6 Wind Erosion

Wind erosion has been reported in various environments in Belgium: agricultural fields, dune areas, industrialized areas and stockpiles and dumps of various kinds of waste (coal dumps, stockpiles in harbours, etc.). In this overview we describe, wind erosion on agricultural land. In Belgium wind erosion occurs predominantly on the sandy soils in the north-east and in the north-west (Goossens, 2002). These areas show a high risk of wind erosion (Figure 1.30.3). Less affected, but still subject to wind erosion, is the central part of Flanders north of the loess border. The soils in the latter area show a loamy sandy texture, making them less vulnerable to wind erosion than sands.

The majority of data on wind erosion on agricultural land in Belgium are in qualitative form. Unlike the dune areas near the Dutch border and near the coast, numerical erosion rates are not available for Belgian agricultural land. The data that are available are based on inquiries in which both farmers and non-farmers have been involved. They show that between 20 and 50 % of the agricultural fields in the north-eastern part of Flanders are affected by wind erosion (Verpoorten, 1977; Peeters, 1980). According to data provided by Vandebeek (1986), between 60 and 70 % of the farmers in this region are affected by wind erosion on their fields. In the north-western part of the country, wind erosion is generally a lesser important problem, although cases of medium (Lamond, 1978; Parrein, 1980) to very severe (De Meester, 1982) damage have been reported. A reduction in visibility to less than 100 m during wind erosion has been reported in the latter area (De Meester, 1982). Also, on the sandy and loamy sand soils north and east of Leuven, wind erosion has been documented (De Vry, 1980; Vansteenwegen, 1980; Volkaerts, 1980).

Despite the absence of numerical studies, confident estimates can be made of the erosion rates on Belgian agricultural land by extrapolating data from surrounding countries with comparable soils, landscape and agricultural techniques. Studies in The Netherlands and the western part of Germany (Bout, 1987; Goossens *et al*. 2001; Gross, 2002) show that two mechanisms are responsible for projecting soil particles into the atmosphere: wind erosion and tillage activities by the farmer.

During wind erosion, the finest particles (dust) are evacuated from the fields without returning to the surface, whereas the coarse to medium-sized particles (sand, coarse silt) rapidly fall back to the land. The former particles are therefore definitively lost, whereas the latter are mainly redistributed within the field. Based on studies on agricultural land in Germany and The Netherlands (Procé, 1988; Schäfer *et al*., 1990; Goossens *et al*., 2001; Gross, 2002), the rate of soil loss (finest particle fractions) due to wind erosion on the sandy and loamy sand soils in the north-western and north-eastern parts of Flanders can be estimated to vary from <1 to >20 t ha^{-1} yr^{-1}, depending on the number and intensity of the wind erosion events. The rate of in-field redistribution is much higher: the transport coefficient ranges from <1 to >20 kg m^{-1} annually, locally reaching values of 150 kg m^{-1} and more. These rates are comparable to those for water erosion and tillage erosion in the loess belt of central Belgium.

Data on soil particle emission due to tillage activity are sparse, especially for European soils and European tillage techniques. Those that are available show that emission due to tillage is of the order of 4–7 times larger than emission caused by pure wind erosion (Goossens *et al*., 2001). The emission of soil particles by tillage operations is therefore a very important erosion mechanism, which has been neglected for years in the international literature. This process also affects the loess soils in central Belgium, which are only slightly susceptible to normal wind erosion. It also occurs on the more stony soils in the south-east of the country (Ardennes). In Belgium, tillage should be considered the dominant process emitting soil particles from agricultural land.

1.30.4 SOIL EROSION-RELATED ENVIRONMENTAL PROBLEMS IN BELGIUM

The off-site problems of water erosion processes are by far the most important environmental consequences of soil erosion in Belgium. These include muddy floods, retention pond sedimentation, river channel sedimentation and dredging and declining water quality due to sediment and sediment-fixed contaminants. In general, on-site problems related to soil erosion are restricted to the loss of fertilizers, crop damage resulting from root exposure or sediment deposits and hampered accessibility in the fields in case of gullying in the short term, and declining soil fertility in the long term on certain spots in the landscape.

1.30.4.1 Muddy Floods

After intense rain events, mainly in late spring–early summer, many villages in central Belgium are confronted by muddy floods originating from intensively cultivated fields (Verstraeten and Poesen, 1999; Bielders *et al*., 2003) (Figure 1.30.4). In two separate questionnaires to local authorities in southern Flanders (Verstraeten and Poesen, 1999) and Wallonia (Bielders *et al*., 2003), the areal extent of muddy floods was established. If we combine the results of both studies, we can conclude that in the hilly loess area of Belgium, at least 56 % of the municipalities are confronted with runoff from arable land. In the remainder of the country, the occurrence of these muddy floods is much lower or almost absent. Only 28 % of the municipalities in the southern part of the Walloon Region experience runoff from agricultural land, mainly because soils in this region are less erodible compared with the silt loam and sandy silt loam soils in the northern part of the Walloon region and because there is much less cropland (6 %) compared with the northern part (42 %). In the northern part of Flanders, muddy floods do not occur given the less erodible soils (sand and silty sands) and the much lower relief. Around 55–70 % of the municipalities in the central part of the country are affected by muddy floods at least once every 1–5 years, and 15–20 % several times a year. For part of the affected area (southern Limburg; see Verstraeten and Poesen, 1999), every location affected by muddy floods during the last decades was recorded. In this area of 408 km^2, 48 locations suffered from muddy floods, which in combination with a frequency distribution of reported events, yields an average number of locations hit by muddy floods of 4.5 per 100 km^2 per year. This figure will be an overestimation keeping in mind that during one single muddy flood event, several locations are hit simultaneously. On the basis of newspaper reports for the period 1987–97, the number of muddy floods within the area surrounding Leuven (1100 km^2) was estimated at 0.5 per 100 km^2 per year. This figure will be an underestimation as not all muddy floods are reported in the newspapers, and because it was not always mentioned whether a flood event had a muddy character or not. The total number of small-scale flooding events (including the muddy ones) equals 3 per 100 km^2 per year. A conservative estimation of the mean value of muddy flood incidence for the whole affected area in southern Flanders will therefore probably range from 1 to 3 per 100 km^2 per year. Muddy floods are mostly limited to the period May–June when the dominant summer crops such as maize, sugar beet and potatoes, have a low vegetative cover, and when the first intense convective rainstorms occur (Verstraeten and Poesen, 1999). It is during this period of the year that the highest soil loss rates by individual rain events have been reported (Steegen *et al*., 2000; Takken *et al*., 1999; Vandaele, 1997). Following a survey of farmers in the loess belt of the Walloon Region, a strong positive correlation was found between the probability of observing soil erosion in a given field and the presence of summer crops such as sugar beet, potatoes, chicory or maize (Bielders *et al*., 2003).

Muddy floods cause serious financial damage to people and to public infrastructure. For a number of events in Flemish municipalities, damage costs are available (Verstraeten and Poesen, 1999). The mean annual cost to private households at the level of a municipality varies between €10 and 30 ha^{-1} (actualized figures for 2003). A rough estimation of the total annual cost of muddy floods to private properties in the affected areas in Flanders (5500 km^2) thus equals €5.5–16.5 million. Financial damage to public infrastructure (clogging of

sewers, cleaning roads, dredging drainage ditches, etc.) is of the same order of magnitude. The cost will depend not only on the severity of the soil loss, but also on the vulnerability of the housing properties, including housing density in critical areas. Since this varies considerably between Flanders and Wallonia, no extrapolation of total costs to the whole central Belgium was made. Finally, the psychological damage to people that have to face muddy floods on a regular basis should be stressed.

1.30.4.2 River and Pond Sedimentation

Soil loss by water erosion processes is also responsible for elevated sediment loads in river channels, especially in Flanders, since most rivers draining the loess belt flow to the main river in Flanders, namely the Scheldt. Table 1.30.2 lists the available data on sediment delivery to rivers (export from agricultural catchments) and sediment transport rates within river channels (see above).

Much of the sediment is stored within the river channel. The aggradation of the riverbed results in a lower discharge capacity, increasing the risk of flooding and navigation problems, especially in the vicinity of ship locks. Frequent dredging of rivers is therefore needed, yet there is often not enough space to dump the sediment, certainly not when it is polluted with a variety of contaminants (pesticides, nutrients, heavy metals, etc.). In 1999, at least 10^7 m^3 of sediment needed to be dredged from the main rivers in Flanders (excluding the Scheldt Estuary), although there was only space to dump 1.6×10^6 m^3 of dredged sediment (Cauwenberghs, 2000, and personal communication). Total costs for dredging and dumping all this sediment was of the order of €235 million in 2000. Furthermore, the annual accretion rate was estimated at 1.2×10^6 m^3, which corresponds to an annual cost of €32 million (actualized for 2003).

To control flooding, many retention ponds have been constructed during recent decades. Most of these ponds are located on or along a river channel and others are located in dry valleys in agricultural areas. Verstraeten and Poesen (1999) estimated that there were at least 100 flood retention ponds in southern Flanders in 1999. There is no update of this information, but based on the number of planned ponds in 1999, local newspaper reports and communications with government administrations, at present there should be around 200 ponds in Flanders. In the Walloon Region also, many ponds have been constructed. No reliable estimate is available for Wallonia but there could be another 100–200 ponds. All these retention ponds may trap significant amounts of sediment, especially after major runoff events. Because these ponds are relatively small compared with the drainage area, they lose their retention capacity fairly rapidly, sometimes within a few years. Again, dredging is required to retain their function as a flood control measure. Although the first report on sedimentation rates in Belgian ponds was made by Gabriels *et al.* (1985) and dates from the late 1970s, no detailed inventory exists on the amounts of sediment that needs to be dredged annually from these flood control ponds. Recently, Verstraeten (2000) and Verstraeten and Poesen (2001) reported that annual sedimentation rates in 20 ponds spread across central Belgium vary between 0.18 and 9.55 m^3 ha^{-1} yr^{-1} for catchment areas ranging from 7 to 5000 ha. The corresponding annual storage losses range from 3 to 28 % with a mean of 10 %. Using the mean values for the 20 studied retention ponds, the total annual mass of deposited sediment within small ponds in Belgium can be estimated at 200 000–400 000 m^3. Dredging pond sediments costs €10–125 m^{-3}, depending on the degree of contamination. Using a mean value of €60 m^{-3} (based on experience with local administrators), the total annual cost of dredging retention ponds in Belgium may range between €12 and 24 million.

1.30.4.3 Delivery of Sediment-fixed Contaminants to Rivers

Adsorbed on soil particles, pesticides and fertilizers will also enter the rivers. This decreases the water quality downstream, and leads to eutrophication. Little is known about the contribution of soil loss to levels of pesticides in Belgian rivers. For sediment-fixed nutrient delivery, however, some data have been gathered. For small agricultural catchments in central Belgium, sediment-associated P export ranges from 1.8

to 39.7 kg ha^{-1} yr^{-1}, with median values of 4–5 kg ha^{-1} yr^{-1} (Verstraeten and Poesen, 2002; Steegen *et al.*, 2001). Based on frequent sampling and analysis of suspended sediment in the River Dijle near Leuven (700 km^2), Steegen (2001) estimated the total P export at 4.2 kg ha^{-1} yr^{-1}. These figures include point sources and diffuse sources (including soil loss). The minimum contribution of soil loss from agricultural land to P transport by the River Dijle was estimated at 40 %. This corresponds to 1.7 kg ha^{-1} yr^{-1} for the whole basin, but taking into account that this P only originates from agricultural land (55 % of the basin), the P export from agricultural land to the river equals 3 kg ha^{-1} yr^{-1}, a value that compares well with the measurements in the small agricultural catchments.

When the river flow (and the sediment) is mixed with sewage water from households or industry, the sediment becomes heavily contaminated with many other substances (e.g. Zn, As, Cu), which makes the dredging of sediments costly (see above). However, although the sediment originates from soil erosion, the contaminants have a different origin (sewers, industry, households, etc.).

1.30.4.4 On-site Consequences of Soil Erosion in Belgium

In the short term, water erosion is the major process that causes on-site problems. This is certainly the case with ephemeral gullying, which can hamper tillage and harvesting operations. Money and time have to be spent filling the gullies. Together with the runoff and sediment loss, nutrients are being exported. This, however, is in economic terms often negligible (e.g. Hanotiaux, 1978). More important is the crop damage. This includes the loss of small plants and seeds due to rilling and especially gullying (Figure 1.30.5), the

Figure 1.30.5 A summer gully formed in a sugar beet field after an extreme event in the loess area of central Belgium. (Photograph copyright Gert Verstraeten)

exposure of plant roots but also the burying of small plants due to severe colluviation. Pissart and Bollinne (1978) mentioned that after an intense storm event in 1974, up to 50 % of the area with sugar beet in some villages in central Belgium showed some damage. In total, 14 % of the area covered with sugar beet (1038 ha) was so much damaged that the farmers could claim a reimbursement from the Ministry of Agriculture for an amount of BEF 3.5 million in 1974, which in 2003 would correspond to €260 000 (using actual reimbursement figures for sugar beet). They further mention that, on average, 3–5 % of the crop yield in the loess belt is lost annually due to soil erosion. After an extreme rain event in May 2000 in another municipality in central Belgium, more than 25 ha of mainly sugar beet and potato (total affected cropland area of 2000 ha) were completely destroyed and reimbursed for €42 000 (data from local administrators and farmers' organization). If we assume that every year up to 3 % of the total area of sugar beet and potatoes in central Belgium is being destroyed by excessive soil loss and sediment deposition, this would correspond to a total cost of around €5 million.

In the long term, loss of fertile soil and a decline in soil productivity may become important for particular areas where the loess cover is thin. In this respect, soil redistribution due to tillage operations is much more severe. Indeed, on many convexities where soil loss due to tillage is very high, the (infertile) Tertiary sands (underlying the loess deposits) are already being incorporated in the plough layer and crop production figures are less compared with other areas. Furthermore, tillage also redistributes organic carbon and fertilizers, thereby depleting convexities of nutrients (Van Oost, 2003). For a typical field in central Belgium, Van Oost (2003) measured grain yields and he was able to compare the spatial pattern of grain yield with that of tillage erosion rates. On the convexities where tillage erosion rates were maximal, grain yields were up to 30 % lower compared with the average yield, and 48 % lower than the highest grain yields at the footslopes where sediment and nutrients accumulated as a result of tillage.

For the northern part of Belgium, where wind erosion is more important, no information on the on-site consequences exist, although they will certainly be present.

1.30.4.5 Total Costs Related to Soil Loss

A detailed and accurate assessment of all the costs associated with soil erosion in Belgium is probably too ambitious for several reasons: (1) not all the different types of consequences and related costs are known; (2) if they are known, they are often difficult to quantify; (3) even if they can be quantified, it is difficult to do this for the total area affected as the data are not centralised. Using the range of costs we estimated for crop damages, muddy floods and sediment dredging in Flanders, it is possible to provide a crude estimate of the minimum annual costs associated with soil erosion in Flanders. This value ranges from €60 to 95 million.

1.30.5 FARMER AND GOVERNMENT RESPONSE

In Belgium, the Regions are responsible for environmental issues, hence soil conservation policies are different between Flanders and Wallonia. In recent years, Flanders has seen the emergence of a soil conservation policy (for more details, see Verstraeten *et al.*, 2003). In December 2001, the Flemish Government issued a decree concerning 'the subsidy of small-scale erosion control measures to be taken by local authorities', often called the 'Soil Erosion Decree'. This decree provides subsidies to the municipalities in the hilly regions of Flanders to make an erosion control management plan indicating where and what measures need to be taken (an amount of €12.5 ha^{-1} plan area is foreseen). Furthermore, a 75 % subsidy is granted for the implementation of all the approved measures indicated on the management plan. Possible measures that can be subsidized will include mostly small-scale technical control measures such as the construction of small dams and pools or grass buffer strips. Soil conservation measures on fields, for

example conservation tillage, cover crops or grassed waterways, are at present not foreseen in this decree. However, other conservation practices such as sowing cover crops in autumn or sowing grass buffer strips with a width of 5–10 m along rivers and sunken lanes are subsidized as agri-environmental measures (mostly for nutrient management). These practices are now also promoted to farmers in the framework of the erosion control policy. In the framework of European regulations, grants have been available since 2005 through the Flemish Plan for Rural Development for more agricultural practices that reduce soil erosion (i.e. grassed waterways, grass buffer strips, sediment retention dams and pools, conservation tillage and no till). These measures were approved by the European Commission in August 2003. The discussion of a management plan and the selection of fields to implement specific control measures take place in close cooperation with the local farmers. Such a participatory approach has also been applied in several demonstration projects in Flanders during recent years.

In the Walloon region, there exist several agri-environmental measures, such as cover crops or grass buffer strips along rivers or field boundaries, and although they are being implemented by many farmers experiencing erosion problems (Bielders *et al*., 2003), none of these were originally meant to control erosion. Following the example of Flanders, however, the Walloon government is also keen to establish a soil erosion control policy in the near future.

1.30.6 CONCLUSIONS

A variety of soil erosion processes are occurring in Belgium. In the central part of the country, characterized by a hilly topography, loess-derived luvisols and much arable land, water and tillage erosion are dominant. Total water erosion rates, including inter-rill, rill, gully and bank erosion, vary between a few to more than $10\,t\,ha^{-1}\,yr^{-1}$ in many agricultural catchments. Average within-field tillage redistribution rates are of the same order of magnitude. Soil erosion by water, however, is also responsible for sediment supply to rivers and ponds and other environmental problems such as muddy floods. Especially during and after intense storm events, soil erosion and sediment export rates by water can reach several tens of $t\,ha^{-1}$ on a catchment scale. It is these consequences of water erosion that led to the emergence of a soil erosion control policy in Flanders. Historical evidence of soil erosion such as gullies under forest and intense colluviation and alluviation indicates the importance of soil erosion by water in shaping the landscape and confirms the suggested mean rates of soil erosion by water from contemporary process studies. However, from tracer studies it is clear that since World War II tillage erosion has been more important in reshaping the landscape. Soil losses due to crop harvesting (SLCH) is the third most important soil erosion process in Belgium, with a mean value of $1.4\,t\,ha^{-1}\,yr^{-1}$. Mean annual values for SLCH for individual crops (e.g. potatoes, chicory, carrots, sugar beet) can range from 2 to $16\,t\,ha^{-1}$ per harvest, thus being equivalent to erosion rates by water and tillage. In regions where water and tillage erosion are less important (e.g. in the northern part of Belgium), SLCH is probably the most important soil degradation process. Also in the northern part, wind erosion is observed on many fields, yet not many measurements of wind erosion intensity have been performed. Finally, within two smaller areas, several tens of landslides have been identified with mean affected areas of around 5–6 ha. Many of these landslides are still active and cause damage to private and public infrastructure.

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1.31

The Netherlands

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1.31.1 INTRODUCTION

Soil erosion by water and wind occurs in The Netherlands (Figure 1.31.1). However, soil erosion is not a problem on a national scale. The total surface area of The Netherlands is 41 500 km^2, of which 23 500 km^2 was agricultural land in 1996 (Statistics Netherlands). The greater part of the country is flat or almost flat land. Soil erosion by water only occurs on 40 000 ha of loess soils in the region of South-Limbourg, which has a hilly topography. Wind erosion is more widespread than water erosion. Eppink (1982) mentions that wind erosion occurs on 97 000 ha of agricultural land in The Netherlands, of which 75 000 ha is seriously affected.

1.31.2 WIND EROSION

Wind erosion is an active process in The Netherlands (Knottnerus, 1979; Eppink, 1982; Eppink and Spaan, 1989). It occurs on (a) the beaches and coastal dunes along 250 km of sandy North Sea coast, (b) part of the sandy arable soils behind the coastal dunes, (c) arable cut-over raised peat bog soils with a sandy subsoil in the provinces of Groningen and Drente in the north-eastern part of the country, (d) sandy arable soils in the

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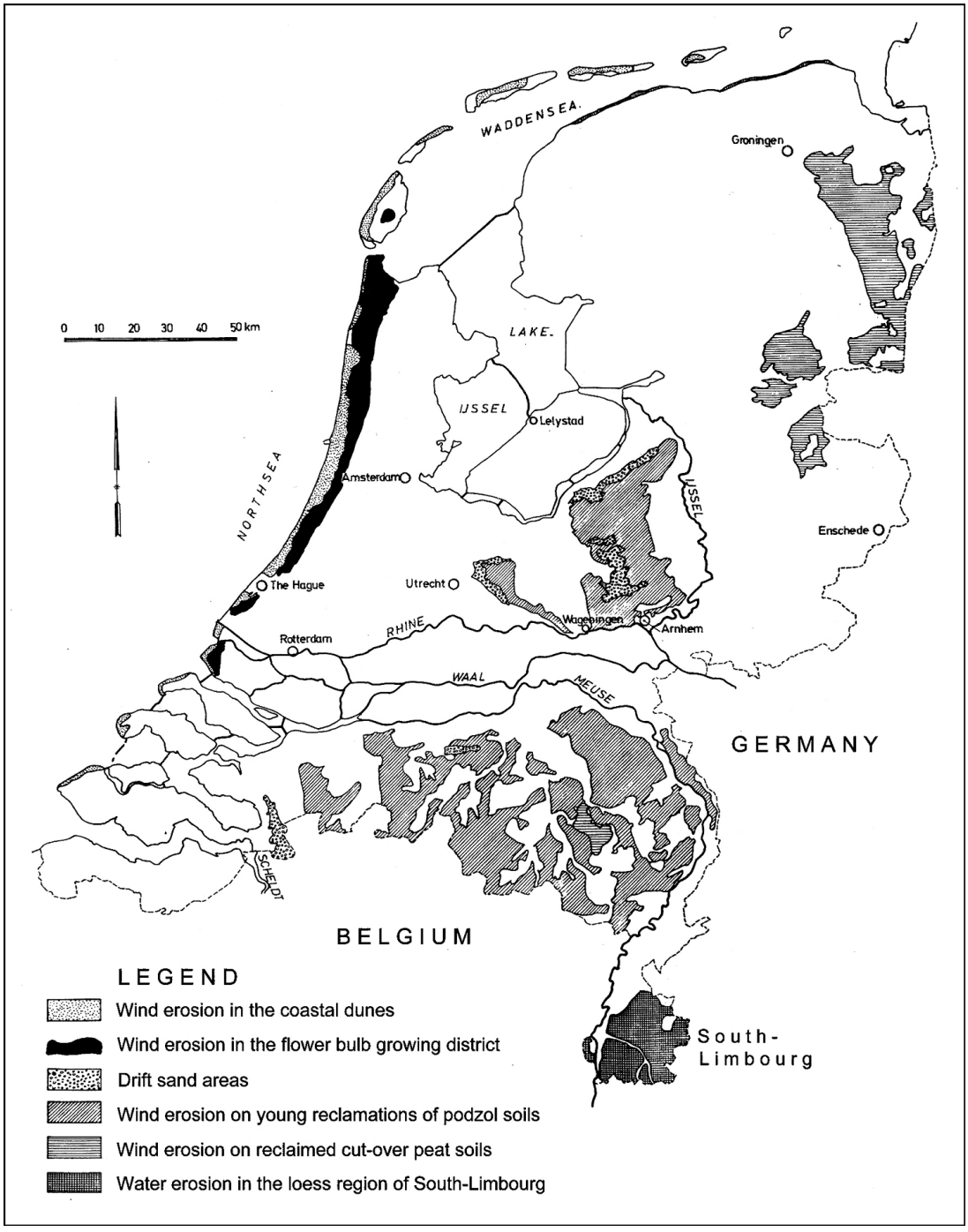

Figure 1.31.1 Erosion hazard map of the Netherlands. [Reproduced from Eppink LAAJ, Spaan WP, Agricultural wind erosion measure in The Netherlands. In *Soil Protection Measures in Europe*, Schwertmann U, Rickson RJ, Auerswald K (eds), Soil Technology Series 1, 1989; 1–13, by permission of Catena Verlag]

province of North-Brabant and along the river Meuse in the province of Limbourg in the south of the country and (e) in aeolian drift sand areas in the centre and north-eastern part of the country (Koster, 1978; Castel, 1991).

Wind erosion along the Dutch North Sea coast has been studied fairly extensively over the past 20 years, with a view to arriving at sound management practices of the coastal zone (Van Bohemen, 1990). Examples of subjects of study in the coastal zone are the development of blowouts in the coastal dunes (Jungerius *et al.*, 1981; Jungerius and Van der Meulen, 1989; Van den Ancker *et al.*, 1985; Pluis, 1993), the testing of an acoustic sediment sensor (Spaan *et al.*, 1991), the sand budget of the foredunes (Arens, 1994), the aeolian transport of beach nourishment sand (Van der Wal, 1999) and the modelling of air flow and sand transport across transverse dunes (Van Boxel *et al.*, 1999; Van Dijk *et al.*, 1999). A main issue in coastal management is stabilization versus so-called 'dynamic' or 'integrated' management of the coastal zone, coastal defence and safety being only one of the management aims. However, as sea beaches and coastal dunes are outside the scope of the present book, they will not be discussed further in this chapter.

In spite of the relatively widespread occurrence of wind erosion on arable land in The Netherlands, little scientific study has been devoted to it, in contrast to water erosion (Eppink and Spaan, 1989; Riksen and De Graaff, 2001). According to Brussel (1980), quoted by Eppink and Spaan (1989), wind erosion occurs on 4–5 days per year every 3–4 years, and on 10 days per year every 15 years. Short-term wind erosion damage to crops was estimated by Eppink (1982) at €9 million €100 ha^{-1} yr^{-1}. This does not include the cost of cleaning roads and ditches and the long-term loss of topsoil and soil productivity. Quantitative data on soil losses by wind erosion on arable land are lacking. According to Eppink and Spaan (1989), measures to combat wind erosion in 1989 comprised the application of straw cover, winter cover crops, grasses, natural soil stabilizers (feed lot manure, manure slurry) and the use of plastic foil.

1.31.3 SOIL EROSION BY WATER

Soil erosion by water is restricted to the region of South-Limbourg in the south of the country, where land use has been mainly agricultural since 1300 AD (Renes, 1988).

Long-term evidence of erosion in South-Limbourg includes truncation of soil profiles, the occurrence of (sub)recent colluvial deposits and the widespread presence of so-called 'lynchets', which are ascribed to deposition of colluvium behind hedgerows on slopes. In 1910, 200 km of lynchets were present in the area. As evidenced by soil profile truncation, the long-term average rate of surface lowering by soil erosion has been of the order of 1 mm yr^{-1} or 15 t ha^{-1} yr^{-1} since the Middle Ages (Bouten *et al.*, 1985). On slopes of 2–8 %, the original A-horizon is removed by erosion. On slopes >8 %, the B-horizon is also removed (Eppink, 1986).

The frequency of occurrence of soil erosion events has increased since the 1970s (Schouten *et al.*, 1985; Schouten and Rang, 1987; Van der Helm, 1988). Soil erosion events with associated off-site effects of flooding and siltation are reported from various sites in South-Limbourg almost every year now. The main forms of damage are (a) (ephemeral) gullying of arable fields and (b) flooding and deposition of mud on arable fields in dry valley bottoms, on roads, in roadside ditches, in culverts and sewers, in the gardens, basements and cellars of houses and in the streets of built-up and residential areas. These are short-term effects that require re-sowing and immediate clean-up action. In a cost–benefit analysis of soil conservation measures in South-Limbourg, Van Eck *et al.* (1995) estimated the cost of the off-site effects at €1.2 million annually. However, no detailed knowledge of the damage of soil erosion and related off-site effects is available for the region. Demands to mitigate erosion mainly come from outside agriculture.

The increased frequency of soil erosion events in recent decades is ascribed to rationalization of agriculture (e.g. increased field size by reallocation of land) and the increased acreage of row crops, especially silage maize. At the same time, the surface area of erodible land (= farmland) has decreased significantly. An erosive effect is also ascribed by some to the increased surface area of built-up land and increased number of paved roads in rural parts of the region, no infiltration of rain water being possible on stone-covered surfaces.

1.31.3.1 Description of the Area

South-Limbourg is a fluvially dissected area of hilly relief, that is dominated by numerous dry valleys. It is part of the drainage basin of the River Meuse. Land surface elevations range from 40 to 321 m. The surface area of undulating terrain is 690 km^2. The centre of the region is at latitude 50° 54′ N and longitude 5° 51′ E. The dry valleys are Pleistocene periglacial relic forms and now act as drainage ways for surface runoff during high-magnitude/low-frequency rainfall events (Kwaad, 1993).

Loess covers 40 000 ha of the region (Van den Broek, 1966; Kuyl, 1980; Mücher, 1986). It overlies coarse-grained Quaternary fluviatile sediments, Tertiary sands and Cretaceous chalk. The thickness of the loess ranges from 2 to 20 m. The loess is mainly Weichselian and was deposited after the main phase of (dry) valley formation. South-Limburg is part of the European loess belt, which extends across south-east England, north-west France, Belgium, parts of Germany and into Poland and Russia.

Luvisols (FAO, 1989) formed in the loess during the Holocene (Stiboka, 1970). The loess soils are highly erodible, owing to their low structural stability and susceptibility to crusting (Kwaad and Mücher, 1994).

The climate of the area is temperate oceanic, with rainfall in all seasons and an annual average precipitation of 750 mm. High-intensity rainfall is restricted to the period April–October (Levert, 1954). The 30-min intensity that is exceeded once a year is 24 mm h^{-1} (Buishand and Velds, 1980). Erosion risk is highest in April–June, when the surface coverage by crops is small and high-intensity rainfall may occur. Prolonged wet weather and rapid snowmelt may cause surface runoff in winter.

1.31.3.2 History of Land Use in South-Limbourg

The natural vegetation of South-Limbourg in the Holocene before the impact of humans was deciduous forest (Janssen, 1960). A first period of deforestation and cultivation included Late Neolithic, Bronze Age, Iron Age and Roman times (1700 BC–300 AD). From about 300 until 1000 AD, forest regrowth took place. Then, medieval deforestation set in, and by 1300 AD the area was completely cultivated and has remained cultivated ever since.

Some data on the history of land use in South-Limbourg are as follows (Jansen, 1979; Philips *et al.*, 1965; Renes, 1988). By 1300, the area was fully cultivated. Over 90 % of the total land surface (69 000 ha) was agricultural land (62 000 ha), of which about 70 % was arable crop land (43 500 ha) and 30 % meadow land (18 500 ha). The dominant arable crops were small grains. Cattle were mainly kept for manuring the arable fields. Grasslands were mostly found in the wet valley bottoms along streams and rivers. The land use situation remained more or less unchanged until around 1900, when the use of artificial fertilizers became common practice.

Land use in 1910, 1960 and 2002 is described in Table 1.31.1. Silage maize was introduced in the region in the 1970s.

Two contrasting effects on soil erosion of the changes in land use between 1910 and 2002 can be distinguished:

1. The 28 000 ha decrease in the area of agricultural land and the 24 000 ha decrease in arable crop land meant a decrease in the total surface of erodible and eroded land, or a decrease in the number of sites or locations where erosion (can) occur(s).
2. The shift from small grains to sugar beet and silage maize on arable land meant an increase in the rate of soil erosion per hectare of (remaining) arable crop land, or an increase in the frequency of occurrence of erosion events on arable crop land sites.

The effect on erosion of the 5750 ha decrease in grassland depended on whether the considered area of grass was turned into arable land or became part of the built-up surface (urban sprawl).

TABLE 1.31.1 Changes in land use (ha) in South-Limbourg

	1910	1960	2002	Increase/decrease
Total surface area	69000	69000	69000	0
Agriculture	60000	?	32000	−28000
Arable crop land	42700	22450	18700	−24000
Small grains	30300	15000	6800	−23500
Potatoes	5340	1900	1200	−4140
Sugar beet	250	2150	4150	+3900
Silage maize	0	0	4150	+4150
Grass and orchards	17250	?	11500	−5750

From 1300 onwards, soil erosion will have occurred widely in South-Limbourg, albeit at a lower rate per hectare than nowadays, owing to the dominance of small grains, small fields and many lynchets. The frequency of occurrence of erosion events on arable land will have been lower in the past (1300–1960) than at present. Since 1910, the total surface area of erodible and eroded land has strongly decreased. At the same time, the rate of erosion per hectare of eroded land has increased, owing to a shift from small grains to row crops and to rationalization of agriculture (e.g. larger fields).

1.31.3.3 Erosion and Conservation Research in South-Limbourg

Early reports of soil erosion in South-Limbourg are scarce. Not until the late 1960s did publications begin to appear on the problem of soil erosion in South-Limbourg: Breteler and van den Broek (1968) on the formation of lynchets by sheetwash and deposition of colluvium behind hedgerows, Kierkels (1971) on the effect of reallocation of land on erosion and Poelman (1971) on factors of soil erosion of loess soils. A decade later, the Landinrichtingsdienst (1983) published a first inventory of 153 flooding locations in South-Limbourg. Schouten et al. (1985) gave a first account of the extent, spatial distribution, rate, causes, damage and control of erosion and Van Eysden and Imeson (1985) of the erodibility of loess soils. Bouten et al. (1985) published an overview of the origin and erosion of loess soils. Van der Helm and Schouten (1986) presented a detailed inventory of 600 erosion sites. Schouten and Rang (1987) drew attention to the costs of soil erosion outside agriculture. Finally, the Provincie Limbourg (1987) gave soil erosion and conservation due consideration in the new regional plan for South-Limbourg.

Until 1985, few quantitative data on the rate of erosion and the cost of the damage were available for South-Limbourg. From the inventory of locations with soil erosion by Van der Helm and Schouten (1986) that covered the whole of South-Limbourg, it appeared that soil erosion and related flooding occur widely in the region. Schouten *et al.* (1985) gave some preliminary and tentative figures of the rate of erosion. They mentioned an average amount of 6.7 t ha^{-1} of displaced soil in rills and gullies during the winter of 1983–84 in 18 first-order catchments in a 1060-ha area (Ransdalerveld) where re-allotment of land had recently been carried out, and 3–30 t ha^{-1} of displaced soil in 6 months on some 30 arable field sites throughout South-Limbourg. The most recent data on the rate of erosion under row crops were collected during an experimental plot study from 1986 to 1993 (Kwaad, 1991; Kwaad *et al.*, 1998). Sediment output on a catchment level was measured during a field project from 1991 to 1994 (De Roo *et al.*, 1995; Van Dijk and Kwaad, 1994b; Van Dijk, 2001).

Immediately following the preliminary assessment of the extent of the erosion problem, research was undertaken in the period 1985–95 that was aimed at the development of measures and procedures to combat erosion. Different farming systems of silage maize and sugar beet were compared on Wischmeier plots, and small nested drainage basins were instrumented. Plot measurements were carried out under natural rainfall and with a large field rainfall simulator. A typical soil loss rate on 6 % and 22-m long plots under natural rainfall in

1987–89 was 16 t ha^{-1} yr^{-1} on fallow plots and 10.8 t ha^{-1} yr^{-1} under conventional maize cropping. This could be reduced to 1.7 t ha^{-1} yr^{-1} by direct drilling of maize in winter rye residue (Kwaad, 1994a). During a comparative plot study of seven cropping systems of silage maize in 1992 and 1993 on 8 % and 22-m long plots, summer soil losses ranged from 0.2 to 4.8 t ha^{-1} and winter soil losses from 1.9 to 4.0 t ha^{-1}, both under natural rainfall. Autumn tillage reduced erosion in winter by 90 %, compared with untilled maize stubble. Applying a surface mulch of finely cut straw (3 t ha^{-1}) after maize sowing consistently gave the lowest soil loss in summer of the seven tested farming systems of silage maize (Kwaad *et al.*, 1998) (Table 1.31.2).

TABLE 1.31.2 Soil loss data from runoff plots, length 22 m, slope 8.5 %, natural rainfall, mean of three replications (g m^{-2})

Cropping system[a]	Winter 1991–92	Summer 1992	Winter 1992–93	Summer 1993
A	28.2	28.7	37.6	147.8
B	58.4	57.5	25.4	292.9
C	19.0	61.6	26.7	127.5
D	79.9	484.8	44.8	300.7
E	34.6	26.6	25.2	110.8
F	405.9	23.6	347.8	20.9
G	84.0	209.1	16.5	179.7

[a]By combining the use of winter rye as a winter cover crop with various times and types of soil tillage, seven cropping systems of fodder maize were devised (Geelen *et al.*, 1996), which were compared in triplicate on 21 plots. Continuous cultivation of maize was applied in all cropping systems for the duration of the plot study (4 years). The cropping systems can be described as follows:

System A: Ploughing, seedbed preparation and drilling of winter rye in October/November after previous maize harvest. Drilling of maize without any form of spring soil tillage in chemically killed winter rye residue in early May (direct drilling).

System B: Ploughing, seedbed preparation and drilling of winter rye in October/November. Maize sown in killed winter rye residue after spring tillage with a Howard paraplough. With this implement, the topsoil is cut loose from the subsoil without disturbing it. The soil is not inverted but lifted by pulling the plough knife through the soil at 25–30 cm depth.

System C: Ploughing, seedbed preparation and drilling of winter rye in October/November. Maize sown in superficially mulched (5 cm deep) winter rye residue.

System D: Only autumn soil tillage (ploughing). No winter cover crop. Direct drilling of maize in spring.

System E: Ploughing, seedbed preparation and drilling of winter rye in October/November. Maize sown in strip tilled winter rye residue. In spring a strip 6 cm wide and 8 cm deep was tilled which was used for sowing. In this way, only 8 % of the total surface area was tilled. Only in the row was a seedbed prepared. A Gaspardo machine was used for the combined tillage and maize sowing operation.

System F: No autumn tillage and no winter cover crop. Maize stubble field in winter. Conventional spring tillage (ploughing and rotary harrowing). Surface mulch of finely cut straw (3 t ha^{-1}) applied after sowing of maize.

System G (reference system): Loosening of maize stubble field in autumn with a cultivator. No winter cover crop. Maize sown after conventional spring tillage (ploughing and rotary harrowing). Since 1990 this is the usual system of maize cultivation in the region. Until 1990, it was usual to leave land untilled during winter under continuous maize growing (i.e. the winter condition of system F). During the trial phase of the development of a maize conservation cropping system autumn tillage greatly decreased winter runoff and erosion (Kwaad, 1994). Therefore, since 1990, local farmers are obliged to carry out autumn tillage on maize fields.

TABLE 1.31.3 Catchment data: rainfall, runoff and sediment output for catchment St Gillistraat 2 (4.8 ha)

Date	Rainfall (mm)	Runoff (m^3)	Sediment output ($t\,ha^{-1}$)
5 June 1992	19.8	5.9	0.087
4 July	5.8	3.6	0.031
5 July	6.2	5.4	0.031
5 July	6.8	11.1	0.061
17 July	7.4	2.9	0.004
13 August	22.6	1.1	0.001
22 November	4.4	24.9	0.035
2 December	8.4	98.3	0.720
11 December	18.4	213.0	0.149
13 January 1993	4.0	24.0	0.128
22 January	11.6	528.3	5.829

Research was also aimed at identifying the mechanism(s) of overland flow generation. In South-Limbourg, Hortonian overland flow, due to surface slaking, crusting and sealing of the structurally unstable loess soils, is generally considered as the prime cause of soil erosion. In the course of the work, however, overland flow was also observed under conditions of low-intensity rainfall. Using various types of evidence, Kwaad (1991, 1993, 1998) and Van Dijk and Kwaad (1996) convincingly showed the occurrence of saturation overland flow in the region. In Table 1.31.3 some results on a storm by storm basis in the St. Gillistraat-2 catchment (4.8 ha) are given. An important erosion event in that catchment occurred on 22 January 1993. During a 10.8-mm storm in 83 min, a runoff percentage of 84.3 % and a sediment output of 28 t or $5.8\,t\,ha^{-1}$ were measured. The return period of the maximum 5-min intensity ($52.8\,mm\,h^{-1}$) of that storm was 4.6 years and that of the maximum rainfall amount in 60 min was 1.8 years (Van Dijk and Kwaad, 1996b).

Outcomes of research for soil conservation in practice were (a) the formulation of conservation cropping systems of row crops, including silage maize, and (b) the Limbourg Soil Erosion Model (LISEM).

Based on the outcomes of research, a conservation ordinance was issued in 1990, which farmers in the region are obliged to observe. This and other regulations are still in the process of amendment by various authorities today (see below).

1.31.3.4 LISEM: Limburg Soil Erosion Model

LISEM is a physically based runoff and erosion model for research, planning and conservation purposes. LISEM simulates runoff and sediment transport in catchments caused by individual rainfall events. The model uses and produces maps based on the freeware GIS PCRaster. The Department of Physical Geography of the University of Utrecht and the Soil Physics Division of the Winand Staring Centre in Wageningen cooperated in the development of this model, assisted by experimental field work of the University of Amsterdam and the Limburg Waterboard (De Roo *et al.*, 1995).

Processes incorporated in the model (Figure 1.31.2) are rainfall, interception, surface storage in micro-depressions, infiltration, vertical movement of water in the soil, overland flow, channel flow, detachment by rainfall and throughfall, detachment by overland flow and transport capacity of the flow. For a detailed description of the processes incorporated in the model the reader is referred to De Roo *et al.* (1996a), Jetten and De Roo (2001) and the website http://www.geog.uu.nl/lisem. LISEM can be applied on small fields and in catchments of up to $10\,km^2$ using time steps of 5–60 s. A sensitivity analysis and validation are presented in

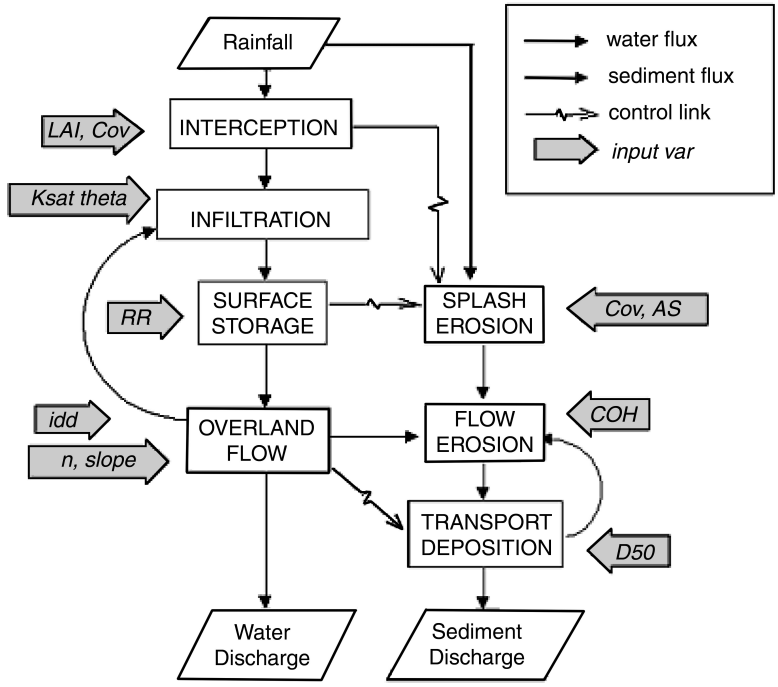

Figure 1.31.2 Flowchart of LISEM. The left-hand column shows the hydrological processes, and the right-hand side the erosion processes calculated per grid cell. Main variables: *LAI* = leaf area index; *Cov* = ground cover; *Ksat* = hydraulic conductivity; *theta* = initial moisture; *RR* = random roughnes; *n* = Manning's *n*; *AS* = aggregate stability; COH = cohesion; *D50* = particle size

De Roo *et al.* (1996b) and Jetten *et al.* (1998). Major conclusions are that the quantitative results of the model are strongly influenced by the knowledge of the spatial and temporal variability of soil moisture content and hydraulic conductivity in the catchment. Examples of use and details on the spatial prediction strength of the model can be found in De Roo (1996), Takken *et al.* (1999) and Jetten *et al.* (2003).

During the LISEM project, all land use types present in the area have been monitored: grassland, winter wheat, winter barley, sugar beet, potatoes and maize. On special trial fields, the influence of 'mulching' and direct sowing has been measured. Variables included in the monitoring were soil cover by vegetation, leaf area index, crop height, random roughness, soil physical parameters, soil texture, aggregate stability and soil cohesion. Thus, a large database has been created on the monthly variation of these variables.

Particular attention is paid in the model to agricultural features: drainage by tillage direction, influence of tractor wheelings, small paved roads, ditches, grass strips and grassed waterways. Because LISEM was designed to model the effect of field level conservation measures such as grass strips, mulch application and changes in crop rotation, one of its more advanced features is the ability to cope with grid cells that consist of different surface types. For each surface type (a particular crop, compacted wheeltracks or crusted parts) a parallel Richards infiltration system is used. The differences in infiltration in a grid cell produce a weighted average of surface water available for runoff. The surface roughness also plays a large role: it determines not only the surface storage but also the hydraulic radius. Current developments in LISEM are the simulation of runoff losses of nitrogen and phosphorus in solution and suspension and the incision and development of ephemeral gullies (Jetten *et al.*, in press).

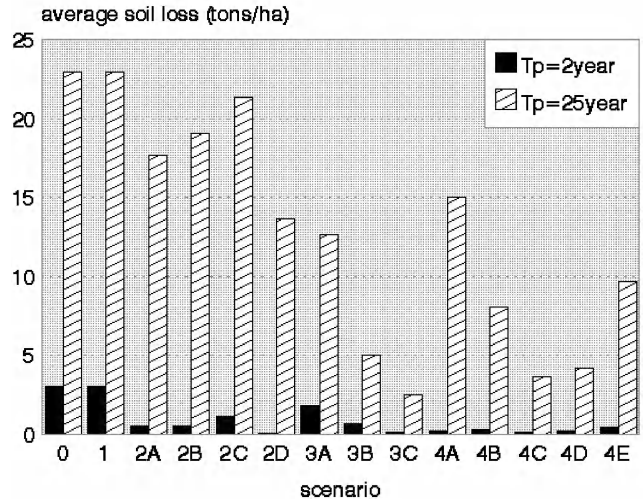

Figure 1.31.3 Effects of 14 scenarios on total event soil loss in the St Gillisstraat drainage basin (Ransdaal, The Netherlands, size 40 ha) for summer storms (20-min duration) with return periods of 2 and 25 years. Scenario 0 is the actual land use in 1990; scenario 1 is the land use in 1993; scenarios 2 are different tillage techniques; scenarios 3 are conservation measures such as field buffer strips and grassed waterways; scenarios 4 are combinations of 2 and 3. Tp = return period

The agricultural features play a large role in Limburg: in certain seasons more than 25 % of the area of agricultural fields consists of compacted wheel tracks with a low infiltration capacity, whereas paved roads make up 2–3 % of the surface area. Moreover, these tracks may influence greatly the connectivity in a catchment. During the LISEM project, it was estimated that roads and wheel tracks may be responsible for 10–25 % of the runoff in a catchment.

LISEM was used to calculate a number of land-use scenarios using summer and winter design storms of 2 and 25 years recurrence time (Figure 1.31.3). The scenarios encompassed different degrees of conservation measures: mulching, cover crops in winter and the application of grass strips on fields with certain slope angles. The results of the scenarios are still used in the present day analysis.

Currently, LISEM is used on a regular basis by the Waterboard to simulate the effect of changes in land use, the application of conservation measures or the design of water retention buffers. Since LISEM produces raster maps with the spatial distribution of erosion and deposition patterns, the effect of different within-field conservation methods can be compared (Figures 1.31.4 and 1.31.5). The south of Limburg is at present (2003) undergoing a major land reallocation operation and the local government is constructing more than 200 water buffers for water and sediment retention. The buffers have a slow-release system and are designed in such a way that they will be filled up in one 25-year event and empty in 24–48 h to the nearest ditch or waterway. At the same time, the government is trying to introduce field-level conservation measures (such as grass strips), for which an elaborate point system is constructed in cooperation with the farmers.

1.31.3.5 Policy and Regulations to Combat Erosion

The objective of soil conservation in South-Limbourg, as elsewhere, is to reduce soil loss and related damage to ‘a level that is acceptable to society’. No soil loss tolerance is specified for the region in terms of an acceptable average long-term rate of soil loss in $t\,ha^{-1}\,yr^{-1}$. Instead, recurrence intervals of 10 years (for rural

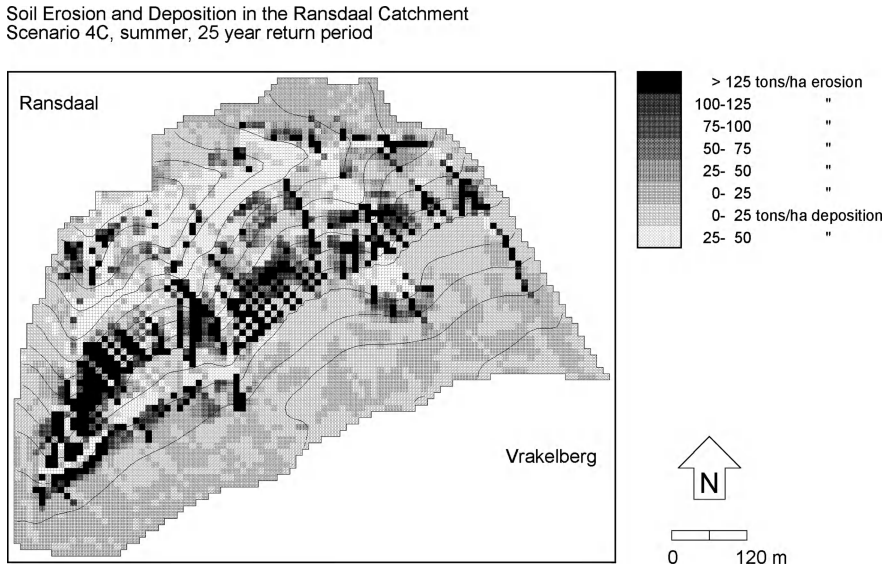

Figure 1.31.4 Soil erosion and deposition in the St Gillisstraat drainage basin (Ransdaal, The Netherlands) for a scenario with field buffer strips and grassed waterways

areas) and 25 years (for residential or built-up areas) are mentioned for erosion events that should be effectively prevented. This includes the cumulative damage of all smaller and more frequent events than once in 10 or 25 years. Farmers consider it their responsibility and obligation to provide a level of protection on their land that is equal to the protection that is provided by small grain, e.g. winter wheat. The farming

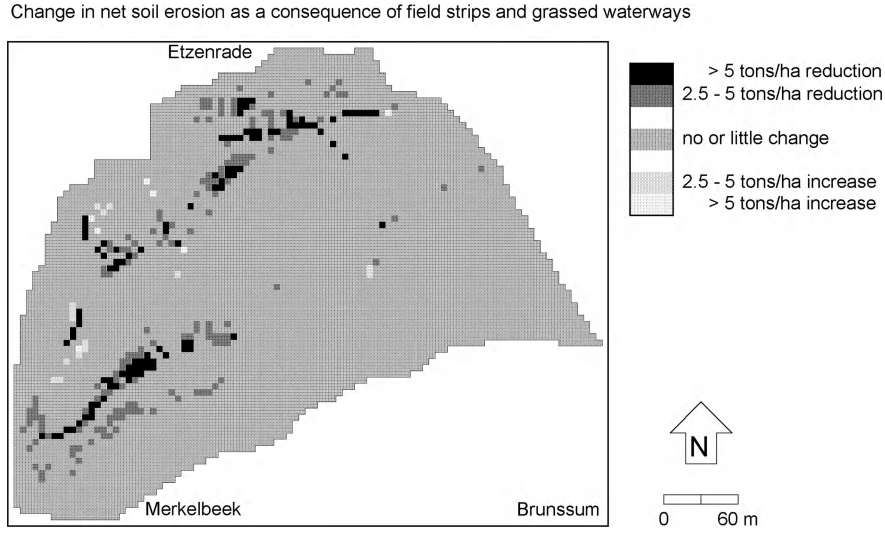

Figure 1.31.5 Change of net erosion as a consequence of one of the land-use scenarios in Figure 1.31.3 compared with the actual land use of 1993

community feels that it cannot be held responsible for damage that occurs in spite of conservation measures having been taken by them that provide a level of protection similar to that of small grain. They consider such damage a calamity that exceeds a conservation effort that can be reasonably expected from them. The level of protection offered by small grain is now the goal of farm conservation plans (see below).

In 1990, a first conservation ordinance was issued in South-Limbourg, in which generic land-use measures were specified to be followed by all farmers in the region. In later years, the ordinance of 1990 proved not effective enough and was revised several times. As of 1 July 2003, the latest revision is in force. On a higher level, additional measures are taken by the municipalities and the Waterboard. Moreover, a site-specific approach was introduced to complement the generic approach. Soil conservation in South-Limbourg is now characterized by (a) a multi-level approach (farmers and municipalities) and (b) the application of generic measures and site-specific measures. Site-specific measures apply both to individual farms and to certain locations in the landscape where erosion and flooding constitute a recurring problem. Actors that must carry out the conservation work are the farmers, the municipalities and the Waterboard. Soil conservation must also be given due consideration in plans for spatially rearranging parts of the region (re-allottment of land).

Ideally, the implementation of conservation measures should be preceded by and based on a cost–benefit analysis. A rigorous cost–benefit analysis, however, is hampered in most cases by a lack of sufficient 'hard' data on the cost of short- and long-term, on- and off-site damage of soil erosion. A slightly different approach to conservation planning, that is followed in South-Limbourg, is to specify a certain return period of events that must be prevented (e.g. 10 or 25 years), and to model the erosion of the 10- or 25-year storm under different land-use scenarios with LISEM, without exactly knowing the cost of the damage of the 10- or 25-year event. Kraak and Van Oorschot (1998) solved the problem of not having sufficient knowledge of the on- and off-site damage of soil erosion as follows. They introduced the cost-effectiveness of a conservation measure, which is defined as the annual cost of the measure per ton reduction of modelled soil loss from a catchment during the 25-year storm. The modelled soil loss reduction is a surrogate variable or index for the cost of the damage that would occur if the measure is not taken. Kraak and Van Oorschot (1998) placed the break-even point between costs and benefits at €140 per ton reduction of modelled soil loss. They advise against measures that cost more than that. It should be remembered in this context that land-use scenarios, which provide sufficient protection against the 25-year event, also curb the (cumulative) damage of all smaller and more frequent events than those that occur once in 25 years.

The main points of the 'Conservation Ordinance' that all farmers in the region are obliged to follow are as follows:

- to perform a post-harvest tillage operation to a depth of 20 cm or more;
- to remove or erase tractor wheelings after the sowing of silage maize or sugar beet, unless direct drilling is used;
- to apply a green manure crop after the harvest of maize or small grain, unless sufficient straw remains on the field that is not worked into the soil by post-harvest tillage;
- to construct a water-retaining barrier of at least 3 m width at the lower end of fields with erodible crops;
- on slopes of 2–5 %, to restrict the field length of an erodible crop to 400 m, or apply one of the techniques of direct sowing, mulch sowing or straw cover after sowing;
- on slopes of 5–18 %, to restrict the field length of an erodible crop to 300 m, or apply one of the techniques of direct sowing, mulch sowing or straw cover after sowing;
- on slopes >18 %, only grassland is allowed.

Instead of applying these generic measures, individual farmers are allowed to draft a conservation plan that is geared to the specific conditions on their farm. An individual farmer múst make a conservation plan for their farm when they want to convert existing grassland that is located in a 'problem area', into arable land.

'Problem areas' are areas with recurring erosion damage that are designated as such by the authorities. Because grass provides the maximum attainable level of protection against erosion, the maintenance or introduction of grassland at strategic points in the landscape is considered an important instrument of soil conservation in the region, especially in a 100-m zone upslope of residential areas and roads. Judgement of the effectiveness of individual farm plans is based on a point scoring system. A detailed list of conservation measures with scores is presented to the farmers to choose from. A score of at least 40 points per hectare should be made. This corresponds to 80 % of the level of protection that is provided by winter wheat (50 points per hectare). Base reference of the scoring system is permanent grass land (100 points per hectare). For fields within 100 m upslope of buildings and roads, a score of 100 points per hectare must be attained.

Municipalities and the Waterboard are responsible for the maintenance and/or construction of (a) linear elements in the landscape (lynchets, grass buffer strips), (b) grassed waterways, (c) grass berms along roads and (d) retention basins for water and sediment.

In addition to the application of generic measures by all farmers in the region and conservation plans for individual farms, site-specific measures are devised for locations where soil erosion and related damage (flooding, deposition of mud) are known to constitute a recurring problem from year to year. The measures for these acknowledged problem areas or erosion hot spots are carried out by the water board and the involved municipalities and farmers (concerted action).

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1.32

Luxembourg

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1.32.1 THE PHYSICAL GEOGRAPHY OF LUXEMBOURG

The landscape of the Grand Duchy of Luxembourg can broadly be subdivided into two main regions (Figure 1.32.1); the Oesling, which is the northern part with a substratum of Devonian slates, phyllites and quartzites (Lucius, 1948) and the southern Gutland, with a substratum of varying Mesozoic sedimentary rocks (Figure 1.32.2) (Lucius, 1950).

The Oesling belongs geologically to the Ardennes–Eifel–Hunsrück massives, in which Devonian and Carboniferous rocks were folded during the Variscan orogenesis. The current Oesling landscape is a remnant of a large planation surface (500–550 m above sea level), with wide shallow valleys in the northwestern part, but increasingly dissected towards the south and south-east, by the Sûre river and is tributaries. This incision started as a result of uplift during the Pliocene, continued during the Quaternary, related to the Alpine orogeny and an active mantel plume with its centre in the Eifel region (van Balen, *et al.* 2000). Owing to this differential uplift, the incision is larger in the east than in the west of the country. The deep incisions reach height differences of up to 250–300 m. Several levels of terraces along the major streams document stages of dissection.

Geologically, the Gutland is situated at the north-east border of the Paris basin and can be characterized as a cuesta landscape and has consequently a very different character. A sequence of several cuestas is present, related to outcrops of resistant Triassic and Jurassic sedimentary strata, which dip slightly inwards to the centre of the Paris Basin. In between the more resistant dolomite, limestone and sandstone formations, less resistant lithologies are present of which the Keuper marls are the most important. The most prominent cuesta is the one developed in the Lower Liassic strata (Luxemburger Sandstone cuesta). The highest parts in the cuesta landscape rise to altitudes of about 400 m, and cover large slightly sloping plateau-like areas, dissected by

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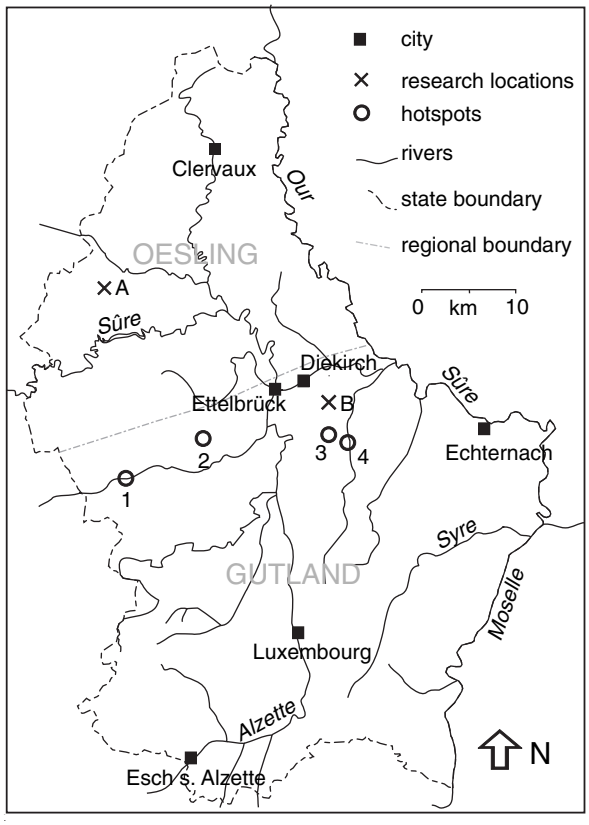

Figure 1.32.1 General map of the Grand Duchy of Luxembourg, showing main rivers and towns, the 'hotspots' discussed in the text, experimental stations and the main regional division. 1, Redange; 2, Vichten; 3, Nommern, 4, Larochette; A, Haartz/Birbaach catchments near Wiltz; B, Schrondweilerbaach, Mosergriecht and Keiwelsbaach catchments

small rivers. In the south of the country, resistant younger Jurassic strata outcrop in two more important cuestas: the cuesta of the Macigno sandstone (medio-liassic cuesta) and the Dogger limestone cuesta (bajocien cuesta) (Désiré-Marchand, 1985).

Luxembourg has a temperate humid climate with an annual rainfall ranging between 800 and 1000 mm in the Oesling and the utmost south-west of the Gutland, whereas the rest of the Gutland has precipitation levels from 800 to less than 700 mm towards the East (Ministére de l'Education Nationale 1971). Average annual temperature is around 9 °C in the Gutland region and around 7.5° C in the Oesling, but is also dependent on altitude. Evapotranspiration rates are typically about 400–500 mm yr^{-1} depending on the land use. Rainfall intensities over 60 mm h^{-1} are relatively rare (Lahr, 1964). However, from recent data, it is becoming clear that rainfall amounts increase through time, especially under prevailing westerly winds (Pfister *et al*., 2000).

Land use in the Oesling differs from that in the Gutland. In the Oesling, the high planation surfaces are mainly under pasture, whereas the steep valley slopes are forested with coniferous and deciduous trees. The Gutland area has a much more diverse land use, which is strongly related to substrate and slope. The steep parts of the cuestas are mainly covered with deciduous forests. The dipslopes in between the cuesta borders are

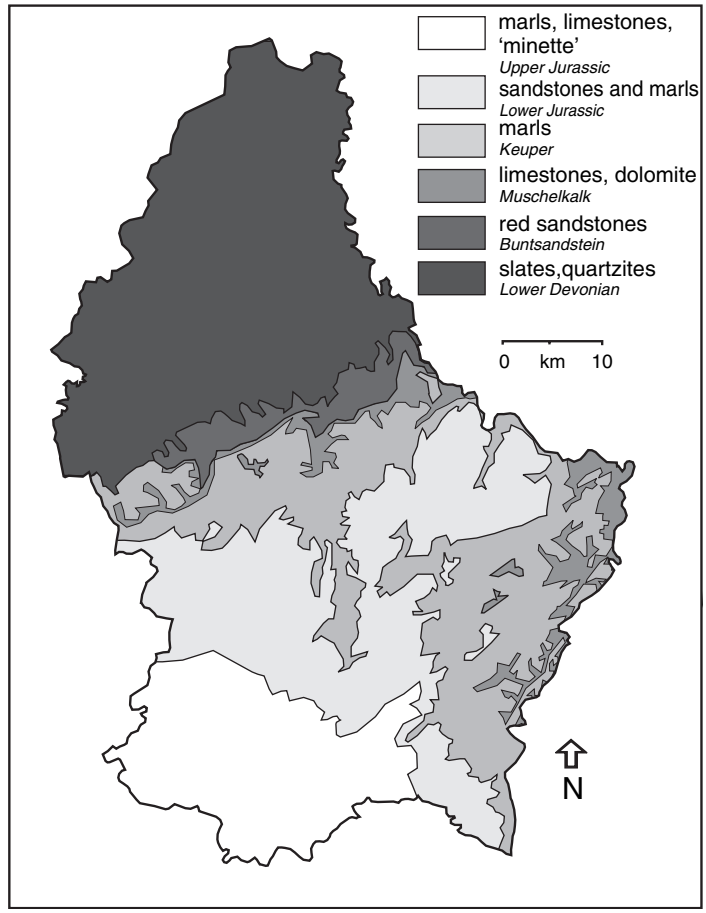

Figure 1.32.2 Generalized geological map of Luxembourg. Emphasis is put on lithology rather than on chronostratigraphic formations

under different types of crops such as wheat and maize, but pastures are also common. Former grassland on heavy marl soils is increasingly used for the cultivation of maize. On the steep slopes along the Moselle river, along the south-east border with Germany, many vineyards are present.

1.32.2 HISTORICAL EVIDENCE OF EROSION

1.32.2.1 The Oesling

Imeson and Jungerius (1974) published a first study on soil erosion in the Oesling near Wiltz and concluded that their were no signs of past soil erosion in the area. Surprisingly, subsequent research in the same region by Kwaad and Mücher (1977) and Verstraten (1978) revealed evidence of rather severe soil erosion from the period before 1400 up to 1800, from pedological, palynological, micromorphological and soil chemical work, for the soils on the plateau tops, steep lower slopes and the dissected valley bottoms, expressed by the presence

of strongly truncated soils, in addition to *Fagopyrum* pollen (buckwheat) in the colluviated topsoil. This crop was not introduced into the area earlier than 1460. They also stated that erosion decreased after reforestation, which started around 1800.

Kwaad and Mücher (1979) subsequently studied the development of colluvial slopes under arable fields. They concluded that most of the colluvium studied under arable land had a late medieval or younger age, again because of the presence of *Fagopyrum* pollen. In another paper Kwaad (1977) states that colluviation rates were on average $0.8\,\mathrm{mm\,yr^{-1}}$ between 1400 and 1800 and have declined to $0.6\,\mathrm{mm\,yr^{-1}}$ since then.

Riezebos and Teunisse (1992) observed that the water–sediment regime in streams must have changed in the early Sub-Atlantic period (3000–2000 BP). It is marked by an increase in fines (fine silts and clays) in the alluvial deposits, which are almost absent in the underlying alluvial sediments. They relate this to increasing activity of humans, whose influence released such large amounts of regolith material to the river system that the transport capacity of the streams could not cope with it. Consequently, this choked the valley bottoms with sediment, in which after the reforestation around 1800 the current streams have incised. The change in sediment availability occurred together with a change in the hydrological regime, expressed by a change from groundwater/subsurface flow dominance to overland flow dominance.

1.32.2.2 The Gutland

As in the Oesling, the history of erosion in the Gutland is strongly related to the history of land use. Evidence of soil erosion is seen in the occurrence of truncated soil profiles, colluvial deposits on foot slopes and sediments trapped in closed depressions (mardellen) and active formation of levées and sedimentation along the main river courses. Soil profile truncation is common in the marl areas, especially on Steinmergelkeuper marls under agricultural land use. Whereas soils under forest with gentle slopes typically show a normal profile development with a clear B horizon, the profiles on the agricultural fields often show only a thin Ap horizon directly on top of the C material or even directly on the weathered shards.

Jungerius and Mücher (1970) analysed the erosion rates for the several characteristic slope elements of the Lias cuesta by studying the concentration of specific volcanic minerals related to the Laacher See eruption (11 800–11 000 BP) in the topsoil. They concluded that, despite the clayey texture of the Ariëten strata on top of the Luxemburger Sandstone, these dipslope areas connected to the top of the cuesta were least affected by soil erosion. The concave part of the cuesta slopes, developed in the marly Steinmergelkeuper, showed the highest erosion rates since the Allerød. Jungerius (1980) subsequently extended these results by quantifying the surface lowering for the Keuper marls to 44–57 cm after the Allerød period. Jungerius and van Zon (1982) discussed slope development on the Lias cuesta in relation to the presence of a protective litter cover at the surface. On the marly Keuper sections of the slopes this cover is less well developed and hence the resulting splash erosion on the bare areas promotes increased surface lowering in comparison with the overlying Liassic strata.

The influence of humans is clearly demonstrated in the colluvial and recent fluviatile sediments indicated by charcoal and the characteristic grey–brownish colours of colluvia. Holocene fluviatile sedimentation in the valley bottoms may reach a thickness of several metres and is still continuing at present, as demonstrated by the presence of a well-layered levée of 75 cm thickness currently being deposited on a paved road dating from the 1960s near Reisdorf.

Historical erosion and deposition rates were investigated by Poeteray *et al*. (1984) for several mardels in the Gutland area. Mardellen, closed depressions for which the origin is still not satisfactorily clarified, have been acting as traps for sediment and pollen. Several cores were investigated and some date back as far as the Roman period. The palynological record was dated against ^{14}C determinations, and known introduction dates of plant species, especially *Fagopyrum* and *Picea* (fir), the latter being introduced

around 1800. From these deposits the surface lowering rates were calculated, showing two clear peaks between 1200 and 1350 (0.086–0.215 mm yr^{-1}) and between 1460 and 1600 (0.100–0.279 mm yr^{-1}), coinciding with two periods of increasing human intervention in the area. After the 16th century sedimentation rates declined (0.055–0.156 mm yr^{-1} from 1600 to 1800 and 0.020–0.054 mm yr^{-1} after 1800) as a result of decreased land use due to the introduction of potatoes, which crop gives a higher calorific yield per hectare, leading to land abandonment and because of reforestation by the end of the 18th century (Poeteray *et al.*, 1984).

Van Hooff and Jungerius (1984) published an extended overview of the occurrence of soil truncation and colluvial deposition for 11 watersheds in the Gutland with marl substratum. This study covered an area slightly larger than 38 km^2. Only 5.8 % of the area showed complete soil profiles, 44.0 % showed truncated profiles, 46.4 % showed colluvial deposits and 3.7 % was occupied by alluvial deposits, with an average soil truncation of 55 cm for the whole area. This is a conservative measure, as in places where the whole solum was lost the total degree of truncation cannot be established. The figure fits well within the values calculated from the presence of volcanic minerals (Jungerius, 1980). They also demonstrated that there is a clear relation between land use and the occurrence of colluvium, the latter being more prominent on lands under agricultural use, but that also geomorphological properties of the catchment are important.

All along the Luxemburger sandstone cuesta escarpment well-developed gullies exist in marl substratum. Their origin can be natural, indirectly human induced or directly made by humans. Some of them are created by hauling of wood taken from the steep forested cuesta escarpments. These, up to 3-m deep gullies, normally cross the steep slopes in an inclined way like a steep road, whereas the natural gullies are developed perpendicular to the contour lines. The latter can be found both under forest but also under grassland. It might well be that some of these gullies originate from periods of historical deforestation or that they are related to periods of overgrazing, as these zones often were used by the local population (‘common grounds’) for grazing and the collection of fuel wood. However, some of these gullies, especially under grass cover, have also formed recently, during extreme rainfall events causing local slope destabilization.

1.32.3 CURRENT EROSION RATES

1.32.3.1 The Oesling

Current erosion rates are not very well known for the entire region as no permanent erosion stations are established. The soils in the Oesling are mainly developed in slates and phyllites, which show relatively good infiltration rates, and hence current erosion rates on the planation surfaces seem to be relatively unimportant (Imeson and Jungerius, 1974; Kwaad, 1977; Kwaad and Mücher, 1977), as indicated by studies carried out in the region around Wiltz. The only relevant surface erosion process under forest at present is splash erosion, in places where the soils becomes bare, frost action and creep (Imeson, 1976, 1977; Imeson and Kwaad, 1976; Kwaad, 1977). The presence of bare soils and hence increased erodibility are almost completely related to faunal activity such as the burrowing activity of earthworms and moles. All authors come to the conclusion that current natural surface erosion processes show low ‘natural’ rates for the forested areas. Kwaad (1977) indicates an average colluviation rate of 0.6 mm yr^{-1} since 1800. Imeson and Kwaad (1976) measured splash erosion rates of 0.0702–0.0751 t ha^{-1} yr^{-1} soil loss on the steep hillslopes and a delivery of 0.0102 t ha^{-1} yr^{-1} to the stream. Chemical denudation rates under semi-natural forest are also very low (0.0623 t $ha^{-1}yr^{-1}$; Verstraten, 1977).

On the arable fields, tillage erosion may be an important factor and it should be comparable to rates observed in the adjoining Belgian and German parts of the Ardennes and Eifel.

Figure 1.32.3 Fresh gully developed in slope deposits on marls in a fallow field as generated during an intense storm on 10 June 2003, near the village of Nommern

1.32.3.2 The Gutland

Most of the erosion studied and observed is again in the areas of the Keuper marls in the Gutland. For the agricultural areas, less detailed data on soil erosion are available than for the forest areas. The general presence of truncated soil profiles and associated colluvia (van Hooff and Jungerius, 1984) with no sign of renewed soil profile development indicates ongoing soil erosion resulting from agricultural practices, such as tillage, and vegetation removal. Some measured sediment yields for watersheds under agriculture exist. For the Mosergriecht (mainly Keuper marl substratum), 1.474, 2.660 and 2.457 t $ha^{-1}yr^{-1}$ of suspended load were calculated for 773, 1016 and 946 mm of annual precipitation in the period 1978–80 (Imeson and Vis, 1984a; van Hooff and Jungerius, 1984) equalling surface lowering values between 0.10 and 0.16 mm yr^{-1}.

Agricultural land use for this catchment was 19 % ploughed and 81 % under pasture for 1978. No data on tillage erosion are known, but again these should be comparable to values for similar types of landscapes from south-east Belgium or north-east France.

On some occasions rill and gully erosion occur. Generally these can be found on slopes with low vegetation cover (just ploughed, fallow field and fields with maize crops). Figure 1.32.3 shows the result of gully formation on a fallow agricultural field on marls covered with sandstone and marl-rich slope deposits. The gully was formed during one extreme rainstorm in June 2003 near the village of Nommern (rainfall intensity: at least 8.8 mm in 30 min at the station of the Lycée Classique de Diekirch; this is approximately 10 km away from the site and rainfall intensity shows large spatial variability, also due to orographic effects). The sediment yield of this event affecting only one field was estimated at 15–18 t ha^{-1}, but is known that this site and other nearby slopes have had earlier problems with gully formation up to more than 1.5 m deep. The same storm created a mudflow in the village of Larochette generated on a steeper field, where recently hedges had been removed, probably because of lower subsidies for hedge maintenance.

Richter (1991) discusses the problems of erosion on steep vineyards along the Moselle valley. He notes that cultivation techniques created a dramatic increase in erosion and destabilization of slopes, which necessitates the application of new control measures. Walter (1981) gives an estimate of the annual soil loss, at between 0.1 and 1.0 t $ha^{-1}yr^{-1}$, depending on the slope and the stoniness of the soil.

For the forested Keuper marl areas, more data are available. Current erosion rates for small catchments are given by several authors (Imeson and Vis, 1984a; Van Hooff and Jungerius, 1984; Duysings, 1987). According to these authors, these small watersheds under forest show suspended solid outputs to be a factor 2–5 lower than for the agricultural watersheds on the same substratum, depending on the watershed studied. The most detailed one studied, the Schrondweilerbaach showed values of 0.765 t ha^{-1} yr^{-1} (equalling approximately 0.05 mm of annual soil surface lowering) for the period 1979–81 (Duysings, 1987; Table 1.32.1). The values of slope denudation confirm the average rates given by Poeteray *et al.* (1984) for the last 200 years (0.020–0.054 mm yr^{-1}). The natural erosion here is due to the very specific nature of the substratum, especially the swell and shrink properties of the soil, the type of vegetation and soil faunal activity and the hydrological regime (Hazelhoff *et al.*, 1981; Van den Broek, 1989; Cammeraat, 2002). The most important sources of sediment are the non-incised hillslopes where the process of subsurface erosion of dispersive clay is very important. For partial areas, sediment yield is higher, with values between 2.0 and 1.45 t ha^{-1} yr^{-1} (Hendriks and Imeson, 1984; Cammeraat, 1992). Duysings (1987) established a very detailed sediment budget for one of the streams in the area showing the contribution of different sources and processes to the total budget of a 60.8-ha catchment (Fig.1.32.4). Soil erodibility was studied for different land uses in the same area (Imeson and Vis, 1984b). They found that soil erodibility, determined by rainfall simulator experiments, aggregate stability measurements and splash erosion measurements, was largest for arable farmland, then forest colluvium, undisturbed forest topsoil and pasture with the smallest erodibility. They further found that erodibility showed a strong seasonal variation.

Dissolved solids outputs from the forested Keuper catchments (0.8–1.48 t ha^{-1} yr^{-1}) are larger than the suspended and bedload outputs, which may be related to the presence of evaporites in the marl substratum. However, for the catchments under agriculture, the chemical and mechanical erosion rates are about the same (Imeson and Vis, 1984a; Duysings, 1987).

Van Zon (1980) gives denudation rates for a small forested catchment on the Luxemburger sandstone at the cuesta escarpment. Measured denudation was found to be very low, in total only 0.010 t $ha^{-1}yr^{-1}$ (5.0 mm per 1000 years) for a small catchment. About one-quarter of this total was transported by sediment on leaves, which may well be an important factor in the explanation of the presence of colluvium on forest slopes covered with litter. Van Zon also gives a rate of 0.012 t $ha^{-1}yr^{-1}$ of erosion for the marly Ariëtenschichten under beech forest.

TABLE 1.32.1 Overview of experimental data on soil erosion rates of Luxembourg

Location	Reference	Plot size (km^2)	Measurement period (month/year)	Slope (%)	Soil type (FAO, 1988)	Rainfall ($mm\ yr^{-1}$)	Land use	Erosion type	Soil loss ($t\ ha^{-1}\ yr^{-1}$)
Haartz (Wiltz)	Imeson and Kwaad (1976)	0.169	5/73–6/74	26–70	Dystric Cambisols	880	Deciduous forest	Splash[a]	0.070–0.075
Haartz (Wiltz)	Imeson and Kwaad (1976)	0.169	5/73–6/74	0–9	?	880	Deciduous forest	Splash[a]	0.010
Haartz (Wiltz)	Verstraten (1977)	0.169	11/73–10/75		Dystric Cambisols	880	Deciduous forest	Dissolved load[b]	0.063
Haartz (Wiltz)	Verstraten (1977)	0.169	11/73–10/75		Dystric Cambisols	880	Deciduous forest	Suspended + bed load[b]	0.016
Mosergriecht[a]	VanHooff and Jungerius (1984)	2.37	1978–80	0–14	Cambi-, Luvi- and Regosols	773–1016	Agriculture[a]	Suspended load[b]	1.47–2.46
Mosergriecht	Imeson and Vis (1984a)	2.37	1978–80	0–14	Cambi-, Luvi- and Regosols	773–1016	70.1 % agriculture, 29.9 % deciduous forest	Suspended load[b]	1.11–1.96
Mosergriecht	Imeson and Vis (1984a)	2.37	1978–80	0–14	Cambi-, Luvi- and Regosols	773–1016		Dissolved load[b]	1.106–1.500
Keiwelsbaach	Imeson and Vis (1984a)	0.93	1978–80	0–14	Cambi-, Luvi- and Regosols	773–1016	Deciduous forest	Suspended load[b]	0.299–0.376
Keiwelsbaach	Imeson and Vis (1984a)	0.93	1978–80	0–14	Cambi-, Luvi- and Regosols	773–1016	Deciduous forest	Dissolved load[b]	0.802–1.023
Nommern	Van Zon, 1980	0.085	5/76–4/77	0–57	Cambi-, Luvi- and Regosols	n/a	Deciduous forest	Total denuadation catchment	0.010
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81		Luvisols, Regosols	1050	Deciduous forest	Dissolved. load[b]	1.480
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81	0–57	Luvisols, Regosols	1050	Deciduous forest	Suspended load[b]	0.710
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81	0–57	Luvisols, Regosols	1050	Deciduous forest;	Bedload[b]	0.055
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81	0–57	Luvisols, Regosols	1050	Deciduous forest	Throughflow[a]	0.032
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81	0–57	Luvisols, Regosols	1050	Deciduous forest	Splash/overlandflow[a]	0.313
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81	0–57	Luvisols, Regosols	1050	Deciduous forest	Lateral corrasion[c]	0.170
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81	0–57	Luvisols, Regosols	1050	Deciduous forest	Soil fall[c];	0.155
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81	0–57	Luvisols, Regosols	1050	Deciduous forest	Mass failures[c]	0.056
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81	0–57	Luvisols, Regosols	1050	Deciduous forest	Splash erosion[c]	0.011
Schrondweilerbaach	Duysings (1987)	0.608	11/79–11/81	0–57	Luvisols, Regosols	1050	Deciduous forest	Soil creep[c]	0.003
Schrondweilerbaach	Hendriks and Imeson (1984)	0.00066	1/82–12/82	0–8	Luvisols	828	Deciduous forest	Suspended load[d]	2.00
Schrondweilerbaach	Cammeraat (1992)	0.00098	1/87–12/88	0–10	Luvisols	901	Deciduous forest	Suspended load pipe[d]	1.45
Schrondweilerbaach	Van den Broek (1989)	0.00098	1/87–10/88	0–10	Luvisols	901	Deciduous forest	Matrix throughflow[a]	0.13–0.26

[a]Plot studies on hillslopes.
[b]Output of catchment outlet.
[c]Riverbank processes.
[d]Partial area outlet.

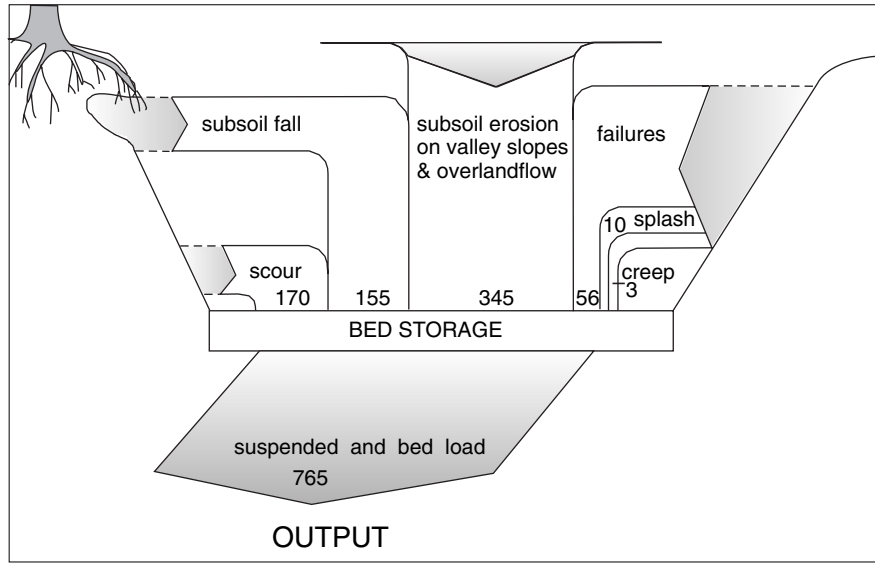

Figure 1.32.4 Detailed sediment budget of the forested Schrondweilerbaach catchment over 1979–81. All values are in kg ha^{-1} yr^{-1}. (Reproduced from Duysings JJHM, A sediment budget for a forested catchment in Luxembourg and its implications for channel development, *Earth Surface Processes and Landforms*, 1987, **12**: 173–184, by permission of John Wiley & Sons, Ltd)

1.32.4 CURRENT PROBLEM AREAS OF EROSION IN LUXEMBOURG

1.32.4.1 General Remarks

In general, it can be concluded that the soil erosion levels in Luxembourg are low to moderate for natural areas. For the agricultural areas this also seems to be the case, except for some problem areas. It is important to note that most of the erosion on marl substratum is related to the dispersion of clays and subsurface erosion processes.

The low to moderate erosion rates are confirming the results provided by the EEA (2003) and by Kirkby *et al*. (2000) for adjoining France. However, at some specific places, on- and off-site erosion problems reoccur and these problems seem to be increasing in extent and also spreading to areas not affected until the last 10–15 years.

Soil erosion in Central Europe increased strongly between 1950 and 1980 after a 150-year period of absence of gullying and soil erosion (Bork, 2003). This increase is related in general to removal of soil conservation measures, increased arable field sizes, changed crop sequences and greater mechanization of tillage (Bork, 2003). It could well be that these changes are also affecting Luxembourg, but that the changes occurred later or that their effect has been revealed later because of a higher resilience to soil degradation of the Gutland soils.

1.32.4.2 The Oesling

It is reported that soil erosion is now also starting to occur in the Oesling, possibly related to the increasing conversion of pastures to fields with maize crops. Natural bogs in valley bottoms, especially in the Oesling, were drained and planted with coniferous trees, reducing natural water retention potential, and wet areas are

also increasingly occupied by built-up areas or infrastructure. This may lead to increasing sediment delivery with increased runoff generation.

1.32.4.3 The Gutland

The main problem areas with regard to on- and off-site problems are concentrated in the Gutland region. They can be separated into (1) areas where excess runoff is generated in combination with sediment transfer in relation to fine textured substrata and (2) areas where large quantities of water concentrate in narrow valleys, which are site-specific problems.

The first type of problems are found in Keuper marl substratum dominated areas, and also on other fine-textured substrata and in areas where land-use changes and management changes and the buffering capacities of the lower areas have declined. Increased runoff generation is often associated with upland erosion and off-site sedimentation. Examples of these are the villages of Vichten and Redange. Here water and sediment are generated from dry valleys spreading into the town centres. These problems have suddenly appeared since 1995 and are now frequently occurring. Analysis of the Redange area revealed that there are several factors that lead to these off-site problems (Cammeraat and Schotel, 1998). Rainfall intensity and recurrence intervals of high daily precipitation rates have been changed, which is a general observed trend over north-west Europe. Changing land use and management practices also have their effects. The increasing areas where maize is grown, replacing former pasture, increases runoff generation. The application of slurry to this sewage tolerant crop strongly affects soil structure. This crop also leaves the fields bare for a prolonged period, making it more vulnerable to erosion. Field surveys also showed that soils were strongly compacted around 30–40 cm depth, reducing the storage capacity of the soil. This creates problems especially during wet periods, including some days with high rainfall amounts. Another common problem is the reduction of natural buffering areas, which is more valid for the Gutland region than for the Oesling, which is far less densely populated.

Runoff and erosion problems are increasingly reported from areas underlain by the Luxembourg sandstone, probably related to the same type of management changes as described for the marl areas. These problems are more prominent where loamy horizons are present in the topsoils, but not exclusively related to such less permeable soils.

Along steep valley sides, such as in the Müllertal and on the Moselle valley slopes, shallow landslides occur frequently after intense rainfall events, blocking roads and producing large amounts of sediments. Soil erosion along footpaths is also reported, especially in the Müllerthal region, which problem has been counteracted by decreasing the extension of the path network (ETI, 2001).

The village of Larochette is a special case as it is located in a narrow valley, with a very limited storage capacity of water along the Ernz Blanche river. The valley bottom is completely built up and the channel cannot cope with increased discharges of the river during high precipitation events. The problem is aggravated by water and sediments coming from several dry side valleys, which also drain large amounts of water during major precipitation events directly into the village, creating problems more than once per year. Currently studies are planned to reduce peak flows and associated sediment deposition.

1.32.5 LEGISLATION/CONTROL MEASUREMENTS

As soil erosion in general is not perceived as a major problem in Luxembourg, only limited attention has been paid to the problem until recently. At the moment there is no national structural programme to reduce erosion risks at potentially vulnerable locations or legislation in relation to the prevention or management of soil erosion. However, a plan to promote ‘sustainable development’ has been adopted by the government (le Plan National pour un Développement Durable; PNDD). It is even marked as a central theme for the Luxembourg

government. This plan recognizes the necessity to protect the environment and natural resources and in which the soil is explicitly mentioned. The section 'soil' in the plan is focused on (a) the maintenance of soil quality and (b) the development of a legal base to protect the soil against chemical, biological and physical pollution. As the problem of soil erosion is increasing in both the Gutland and the Oesling, this is a welcome and necessary initiative to stop and mitigate the adverse effects of soil erosion, which strongly affects soil quality.

In some places, which now frequently encounter the problems associated with upland soil erosion and excess runoff production, retention basins are being built (Redange) or planned (Larochette) as a buffering instrument for sediment and/or water. However, so far no recommendations for planning and land-use management in the upstream areas are incorporated into such measures, which would reduce both the on- and off-site problems. It would be helpful if these considerations were also incorporated into environmental assessments addressing the issue of both on- and off-site aspects of erosion.

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1.33

Britain

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1.33.1 INTRODUCTION

This chapter will review erosion on agricultural land in England, Wales and Scotland. Erosion which affects agricultural land as a result of landslides and coastal and river bank erosion is not covered. Thus we consider accelerated erosion associated with farming and forestry activities. Previous reviews of erosion in Britain include those by Morgan (1980), Evans and Cook (1987), Boardman and Evans (1994) and Evans (1996).

Erosion is of broadly two types: that affecting the upland areas of the west and north and that in the lowlands of the east and south. The former areas have generally more than 800 mm of precipitation per year and land use is largely grassland for grazing. Most arable land is in the east and south where rainfall is from 800 to under 500 mm. Arable land in the UK covered 4.7×10^6 ha in 1999, of which 39% was under wheat and 25% under barley (MAFF, 2000). Most of the wheat is winter wheat planted between September and November.

1.33.2 HISTORICAL EROSION

Major reviews of erosion in the past have been undertaken by Evans (1990a, 1992, in press). Bell and Boardman (1992) include several case studies of past erosion in Britain and Favis-Mortlock *et al*. (1997) model erosion on a South Downs field for the last 7000 years using EPIC (Erosion–Productivity Impact Calculator).

Evidence for erosion in the past shows that it was widespread in the areas where it presently occurs (Evans, in press). Much erosion at the present time occurs on arable land when the ground is bare or partly vegetated. It is influenced by slope steepness and form and soil texture (Evans, 1980, 1993a, 1996, 2002), field size, and

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farming practices (Evans, 1988). Channel erosion occurs on slopes steeper than 2° below convexities and in valley floors. Rainfall thresholds at which runoff and erosion are initiated are surprisingly low, only ca 10 mm, on ground prepared for crops (Evans, 1990b; Boardman, 1993; Chambers and Garwood 2000). Conversely, high-magnitude storms produce little or no erosion if the landscape is well vegetated. The arable landscape is therefore sensitive to erosion at certain times of the year (Brunsden and Thornes, 1979). On winter cereals there is a clear 'window of opportunity' for erosion in the autumn months, which are also the wettest of the year (Boardman, 2003). These principles are likely to have influenced erosion in the past.

There is little evidence for erosion in Britain in the wooded landscapes prior to 5000 BP (Macklin and Lewin, 1993). The beginnings of woodland clearance by Neolithic peoples and the onset of cultivation resulted in colluviation and valley floor alluviation. The Wessex chalklands, East Anglian Breckland and the South Downs were mostly cleared of woodland by 3500 BP. On the South Downs, most of the original loess cover was lost as a result of Bronze and Iron Age cultivation, giving rise to stony, silty soils of less than 20 cm depth in modern times (Favis-Mortlock *et al.*, 1997). By the late 11th century AD, probably about 15% of Britain was wooded and by the late 17th century only about 8% was woodland (Evans, 1993b).

Erosion was most widespread during periods of highest population pressure when cultivated land was extensive and much of the woodland cover had been cleared. For example, alluviation in the upper Thames basin was related to settlement, land use and population pressure (Lambrick, 1992; Robinson, 1992). There is little evidence that colluviation and alluviation were related to climate change (Evans, in press); indeed, Bell (1982) shows that in southern English valleys, dates of earliest deposits vary greatly depending on dates of clearance.

In many upland areas, increased numbers of sheep have influenced erosion rates (Edwards and Rowntree, 1980; Dearing *et al.*, 1981; Tallantire, 1997; Evans 1997; Van der Post *et al.*, 1997). Widespread erosion of peat moors initiated in the mid-18th century is probably related to increasing sheep numbers, accidental and deliberate burning and killing of protective mosses by industrial pollution (Evans, 1996).

In some areas of the country, colluvial deposits indicate that past erosion rates were greater than those of the present, e.g. the Jurassic limestones areas. However, silty soils in particular were much more erodible than their modern counterparts which tend to be stonier (e.g. Favis-Mortlock *et al.*, 1997).

1.33.3 CURRENT EROSION

1.33.3.1 Water Erosion of Arable Land

As a result of the ploughing-up campaign in World War II and post-War agricultural subsidy support, the last half century has seen an intensification of British farming. Many areas not recently cultivated were turned over to arable systems. Heavier and more powerful farm vehicles able to cultivate steep slopes, the loss of field boundaries and the use of power harrows to produce finer tilths (Speirs and Frost, 1985) were factors that increased the risk of erosion. In the mid-1970s, many farmers changed to winter cereals, which became the dominant crop. As a consequence, large areas of the south and east became at risk of erosion in the autumn months (Evans and Cook, 1987). The switch into winter cereals explained the rapid increase in occurrence of erosion on the rolling chalk hills of the South Downs, Sussex (Boardman, 1993). The area of winter cereals peaked on the South Downs in the mid- to late-1980s and around the same time elsewhere in Britain.

Arable areas are subject to water erosion by sheet flow, rilling and ephemeral gullying. A small area is regularly at risk of wind erosion. There is some overlap in terms of risk (Evans, 1990c). There is considerable evidence that overgrazing by sheep is the most important factor in the uplands (Evans, 1977, 1997, 1998a; Phillips *et al.*, 1981; McHugh *et al.*, 2002a,b), but a combination of human trampling, water, wind, mass movements and fire contribute to the loss of soil.

Sheet erosion (or inter-rill erosion) is rarely a significant factor in transporting large volumes of soil within a field. In almost all cases of erosion on arable land rills are present even if they are 'micro-rills' or 'traces' (Colbourne and Staines, 1987). Evans (1990b) suggests that: 'Generally, wash on most soils under arable crops will transport $< 0.3\,m^3\,ha^{-1}\,yr^{-1}$, and this probably applies to all sloping arable fields.'

High-intensity storms on bare sandy soils may be an exception. Boardman *et al.* (1996) note the soil losses of perhaps 20% from inter-rill areas leaving stones on small pedestals. This observation was on a severely eroded bare, maize field on a loamy sand (>80% sand) as a result of 100 mm of rain falling in about 4 h.

The proportion of total erosion due to ephemeral gullying is likely to vary temporally and spatially. On the South Downs in the decade 1982–91, in the years with serious erosion the proportion was around 23% (Boardman, 2003). On clayey soils where slopes may be uneroded, but water concentrates in valley bottoms, ephemeral gullying is more important (Evans and Cook, 1987; Evans, 2002).

1.33.3.2 Wind Erosion of Arable Land

Much less work has been carried out assessing the erosion and impacts of wind erosion than of water erosion (Evans and Cook, 1987; Evans, 1996). In England, wind acts on the sandy soils of the East and West Midlands and East Anglia (see Section 1.33.5), and especially the fine sandy soils, former windblown deposits, of the Vale of York and Lincolnshire. Fenland peat in East Anglia also blows, but as the peat wastes wind erosion becomes less severe and extensive. In Scotland, wind erosion occurs on the sandy soils fringing the Buchan and Banff coast and round to the Murray Firth and those parts of the 'machair' exposed by the plough in the Islands. It appears that when it does occur wind erosion can be more severe than water erosion, but it probably occurs less frequently.

Coastal dunes erode by wind where the vegetation is broken through by holidaymakers either on foot or in their vehicles (Liddle and Greig-Smith, 1975).

1.33.3.3 Upland Erosion

In the uplands, water, wind, frost and animals act together on bare soils (Evans, 1997) and it is difficult to separate out the relative efficiencies of the different processes. Erosion can be extensive on peat and occasionally spectacular, primarily because it has been eroding for centuries (Tallis, 1997; Evans, Chapter 2.11). Before erosion can occur, the land must be stripped of its protective vegetation cover. Bare soil may be initiated by climate induced mass movements or by the actions of mankind. Much erosion is of recent origin.

The survey and monitoring work (see below) carried out by McHugh and colleagues (McHugh *et al.*, 2002a,b) shows that much of the present-day erosion is accounted for by mankind's activities, especially the overstocking of the uplands by sheep (Evans, 1997) and the associated activities of moorland managers to make an income from sheep farming, and also from the shooting of grouse and red deer and from forestry.

Ditches have been dug in the uplands, especially in the peat moors, to drain the topsoil and encourage more palatable and nutritious herbage for sheep. The ditches rarely improve the drainage for more than 1–2 m either side, but they act as a conduit through which rain falling on the land quickly flows across slopes and into streams. Such ditches often lead into seepage hollows and valley heads and may contribute to the destabilization of slopes during large storms, causing large mass movements (see Chapter 2.11). Moors are burnt to provide a variety of vegetation covers and stands of heather of different ages. It is this patchwork of burnt and unburnt moorland in which grouse prefer to live and breed. Such burnt slopes can suffer severe erosion (Anderson, 1986; Alam and Harris, 1987). Roads and tracks constructed across the moors and hills to allow access for shooters are vulnerable to water erosion. Hikers too create paths which erode (Chapter 2.11).

Since World War II, large areas of land in the British uplands have been planted to coniferous forests. Large machines have ploughed up the moors to enable trees to be planted on the better drained ridges, drainage

ditches have been dug and forest roads constructed. All can lead to erosion, as can later grading of roads and the harvesting of trees, especially along river banks.

1.33.4 MONITORING OF EROSION

Water erosion in the farming landscape needs to be monitored at the field scale because small plots give unreliable results which should not be extrapolated (Evans, 1993c, 1994; Boardman, 1998).

Water erosion in England and Wales was monitored at 17 localities in the period 1982–86 (covering ca 700 km^2). Eroding fields were identified on air photographs and field measurements were made of the volumes of rills and gullies. The methodology is cheap and reasonably reliable (Evans and Boardman, 1994). Median, rather than mean, rates are preferred by Evans and Boardman because of the skewed distribution of field-measured erosion data (e.g. Figure 1 in Evans, 1998b, and Figure 3 in Boardman, 2003).

The extent, frequency and rates of erosion for the 1700 fields in the 17 localities were estimated (Evans, 1993a), as were rates of erosion for 20 of the 67 soil associations (Table 1.33.1). Mackney *et al.* (1983) describe briefly the soil associations. The mean maximum area affected by rilling per year was in Nottinghamshire, where 13.9% of the arable land was eroded. Rilling was most widespread on sandy and coarse loamy land. Fields rilled

TABLE 1.33.1 Rates of erosion at 17 monitored localities in England and Wales, 1982–86, and of the soil associations within them with more than 30 eroded fields

	Erosion ($m^3\ ha^{-1}$)				Erosion ($m^3\ ha^{-1}$)			
Locality	Median	Mean	No.	Soil association	Median	Mean	No.	Topsoil texture
Bedfordshire	0.31	0.47	65	411d Hanslope	0.29	0.58	49	Clayey
Cumbria	0.36	1.50	34	—	—	—	—	—
Devon	1.22	1.51	19	—	—	—	—	—
Dorset	0.81	1.35	92	411b Evesham 2	0.74	1.41	63	Clayey
Gwent	0.83	1.43	73	571b Bromyard	0.80	1.29	31	Fine silty
				541a Milford	0.79	2.08	35	Fine loamy
Hampshire	1.33	3.95	59	571i Harwell	1.30	3.52	55	Loamy
Herefordshire	0.68	1.20	89	571b Bromyard	0.67	1.40	68	Fine silty
Isle of Wight	1.52	4.39	141	571g Fyfield 4	1.62	5.54	128	Coarse loamy + sandy
Kent	3.58	4.82	41	—	—	—	—	—
Norfolk East	0.76	1.03	118	551g Newport 4	0.50	1.10	40	Sandy
				541t Wick 3	0.38	1.05	63	Coarse loamy
Norfolk West	0.25	0.85	110	343g Newmarket 2	0.58	1.12	60	Coarse loamy + sandy
				581f Barrow	0.19	0.92	45	Coarse loamy
Nottinghamshire	0.71	1.49	209	551b Cuckney 1	0.72	2.32	191	Sandy + coarse loamy
Shropshire	0.90	2.36	197	572m Salwick	1.25	3.33	36	Fine loamy
				551d Newport 1	0.97	2.55	38	Sandy + coarse loamy
				551a Bridgnorth	0.96	2.76	94	Sandy + coarse loamy
Somerset	2.55	4.69	161	541m S. Petherton	2.10	4.76	114	Silty
				572i Curtisden	1.39	2.58	30	Silty
Staffordshire	0.82	2.43	205	551a Bridgnorth	1.06	3.15	100	Sandy + coarse loamy
				551g Newport 4	0.66	2.60	53	Sandy
Sussex East	0.32	0.62	30	—	—	—	—	—
Sussex West	0.29	0.80	62	343h Andover 1	0.37	1.02	32	Silty

TABLE 1.33.2 Erosion rates in the monitored area, South Downs, 1982–91

Year	1 September–1 March total rainfall (mm)[a]	Median soil loss ($m^3\,ha^{-1}\,yr^{-1}$)	Total soil loss[b] (m^3)	No. of sites
1982–83	724	1.7	1816	68
1983–84	560	0.6	27	7
1984–85	580	1.1	182	25
1985–86	453	0.7	541	49
1986–87	503	0.7	211	34
1987–88	739	5.0	13529	97
1988–89	324	0.5	2	1
1989–90	621	1.4	940	51
1990–91	469	2.3	1527	43
1991–92	298	1.2	112	14

[a]Southover, Lewes raingauge.
[b]Soil loss estimated by measurement of volume of rills, gullies and fans.
Adapted from Boardman and Favis-Mortlock (1993).

on average in most localities at frequencies of 1–3 years, and the mean rates per field were 3 to $> 5\,m^3\,ha^{-1}\,yr^{-1}$ in a small number of localities, but generally were less than $1.5\,m^3\,ha^{-1}\,yr^{-1}$ (Table 1.33.1). High rates in Kent were associated with irrigation of vegetable and salad crops (Boardman and Hazelden, 1986; Evans, 1993a)

Monitoring at a more detailed scale, but over a smaller area, was carried out by Boardman on the eastern South Downs in the decade 1982–91. The methodology was similar to that of Evans in that volumes of rills, gullies and fans were measured in all eroding fields in an area of about $36\,km^2$ of cultivated land (Boardman, 2003).

Erosion rates were again relatively low being less than $5.0\,m^3\,ha^{-1}\,yr^{-1}$ in all years (Table 1.33.2). However, the year-on-year variation was from 0.5 to 5.0. Exceptionally wet autumns such as 1987 give rise to extensive erosion on winter cereal fields with rates of $> 200\,m^3\,ha^{-1}$ on individual fields (Boardman, 1988).

Erosion rates vary with crop type partly because some crops such as maize, sugar beet and potatoes tend to be grown on erodible soils. Mean rates on the latter crops are particularly high although their areal extent is modest. On the other hand, rates on winter cereals are low but they are grown on large areas (Table 1.33.3).

There have been two other schemes to monitor erosion in lowland England and Wales, one for 5 years between 1990 and 1994 (Chambers and Garwood, 2000) and the other for three years, 1996–98 (MAFF/SSLRC, 1998). The earlier scheme covered only a small number of fields (2–13) in a small number of localities (13) covering small amounts of land (29–280 ha). The later scheme visited a large number (ca $260\,yr^{-1}$) of sites based on a grid sample, but of this large number only a few sites eroded in a year so the data provided information only on a national basis, not at a regional or soil association/landscape scale. The amounts of soil eroded were estimated in the field, hence the method of assessment of erosion was similar for all the monitoring schemes, although the areas related to the measurements could vary, from a small catchment within a field to the whole area of land enclosed by a field, for example. The rates of erosion are not dissimilar for the different schemes but because of the different ways in which the projects had been set up, it is not easy to compare the results to or draw the conclusion that water erosion of cultivated land is getting worse in terms of its extent and severity, although it may be inferred from the MAFF/SSLRC data that small-scale erosion has become more extensive. However, again, this may be because these smaller scale events, denoted by flow-lines and 'traces' (Colborne and Staines, 1987), were more specifically targeted in the field survey.

Exceptional storms and their impact on erosion have probably been more studied in Britain (Evans and Morgan, 1974; Evans and Nortcliff, 1978; Boardman, 1988; Davidson and Harrison, 1995; Boardman *et al.*, 1996) than

TABLE 1.33.3 Rates of erosion in soils drilled to different crops[a]

Crop	No. of eroded fields	Mean rate ($m^3\ ha^{-1}$)	Median rate ($m^3\ ha^{-1}$)	National crop area (%)
Market garden and vegetables	102	5.08	1.47	3.1
Maize	26	4.48	1.00	0.4
Ley grass	68	4.09	1.14	4.8
Hops	8	3.92	1.01	0.1
Sugar beet	296	3.04	0.92	4.4
Other[b]	49	2.67	1.07	1.1
Potatoes	171	2.53	1.01	3.2
Kale and other fodder crops	12	2.10	1.41	0.6
Oilseed rape	25	1.92	0.30	5.2
Winter cereals[c]	689	1.85	0.68	60.2
Spring cereals[d]	186	1.75	0.71	13.6
Bare soil/fallow[e]	25	1.61	0.27	1.1
Peas	16	1.21	0.91	1.3
Field beans	6	0.47	0.22	0.9

[a]Data from national soil monitoring scheme, 1982–86.
[b]Crops include soft fruit, root crops for stock feed, strawberries, orchards, linseed, etc.
[c]Dominantly wheat, but also barley and to a lesser extent oats and triticale.
[d]Predominantly spring barley.
[e]Soil surface cultivated but not drilled or rough fallow.
From Hossell and Evans (in press).

in many other European countries. These studies can be valuable because they are an insight into processes that operate over long periods (cf. Bork's work in Germany).

Erosion in the uplands has been surveyed in Scotland (Grieve *et al*., 1995), although little distinction is made between present and past erosion (see Chapter 2.11). Harrod *et al*. (2001) and McHugh *et al*. (2002a) describe a survey of erosion and a 3-year erosion monitoring scheme for the uplands of England and Wales.

1.33.5 EROSION RISK

Quantification of erosion rates measured in the British landscape has led to attempts to model and predict British erosion. The SSEW data proved difficult to use in new or existing models – problems that were discussed by Evans (1990b, 1998b). Indeed, it may not be possible to model risk or occurrence of erosion based on erosion rates, except for small areas (Evans, 1998b). Other models have not worked well, neither a plot-based model (Quinton, 1994) nor one to predict where erosion will occur (Thompson and Beard, 1989). A minimum information requirement modified WEPP model does not adequately predict the extent and severity of erosion as mapped in the field (Evans and Brazier, 2005). However, risk of erosion is indicated in the legend to the National Soil Map (Mackney *et al.*, 1983) and risk of erosion on winter cereals is the subject of a separate mapping and assessment exercise (SSLRC, 1993). Of greater value is the detailed compilation of Evans (1990c) of data based on the National Soil Map, the SSEW monitoring scheme, personal observations and publications which he brings together in an assessment of the 296 soil associations in England and Wales in terms of their susceptibility to erosion risk (Table 1.33.4). Evans shows that a large proportion (36%) of the arable area is at moderate to very high risk of erosion.

TABLE 1.33.4 Soils at risk of accelerated erosion in England and Wales

Risk	No. of soil associations	Area of England and Wales (%)
Very small	108	38.2
Small	109	38.0
Moderate	60	18.0
High	15	4.4
Very high	4	1.5

From Evans (1990c).

Figure 1.33.1 shows the areas in Britain considered to be most at risk of erosion, be it water, wind or upland erosion. This is based on the work of Evans (1990c, 1996) for England and Wales, and for Scotland on the literature describing water erosion in the lowlands (Speirs and Frost, 1985; Watson and Evans, 1991; Kirkbride and Reeves, 1993; Davidson and Harrison, 1995; Wade and Kirkbride, 1998), peat erosion due to overgrazing (Birnie and Hulme, 1990; Birnie, 1993) and relating this information to the national soil map of Scotland and the descriptions of erosion in its accompanying bulletins (MISR, 1982). Throughout Britain there are many narrow coastal dune belts which are susceptible to wind erosion but they cannot be shown on a small-scale map such as this. Also, there are many small units, often of complex shape, vulnerable to water erosion, which too cannot be portrayed at this scale. Many sandy soils which are at risk of water erosion are also vulnerable to wind blows, but to a lesser extent, for instance along the coast of East Anglia and in the west and east Midlands. For England and Wales, because the information base is better, it is easier to identify upland landscapes more at risk of erosion due to overgrazing. For Scotland, it is less easy, partly because of the way in which the soil landscape units are described but also because what could be vulnerable slopes often do not appear as heavily grazed as in England and Wales. Heavily grazed and eroding landscapes in Scotland, such as those associated with the basalt outcrop in the western Highlands and Islands, are often long and narrow in shape and difficult to portray at this scale. In Scotland, many peat units or soil associations with peat within them are noted as having eroded patches within them, but most peat units are of small extent, and are not often noted as being particularly severely eroded.

1.33.6 THE IMPACTS AND COSTS OF EROSION: ON- AND OFF-SITE PROBLEMS

1.33.6.1 On-farm

The removal by water of sufficient soil, fertilizer and crops to make the farmer aware of erosion is rare. Losses in yield caused by rills, gullies and their associated deposits are minimal (Evans, 1993a) and costs attributed to declining yields because of thinning of the topsoil are also slight (Evans and Nortcliff, 1978; Evans, 1995). Farmers with land more vulnerable to wind erosion tend to be more aware of erosion because it is often a high-value crop which is removed (Evans, 1996). Only if the field has to be redrilled or agricultural operations cannot be carried out because of the presence of gullies which have to be infilled is the farmer likely to be aware of the costs of erosion. It is rare that erosion will cost a farmer more than a few hundred pounds per year (Evans, 1995, 1996), and that only in areas vulnerable to erosion. Compared with the present agricultural subsidy of over £200 ha^{-1}, the costs are very small. Loss of riparian land by rivers eroding their banks are likely to exceed £4 million yr^{-1} as this figure was estimated (Evans, 1996) only for the rivers in the Welsh borders (Newson, 1986), and it is known rivers in their 'piedmont' zones are also eroding as they leave the valleys of the Yorkshire Dales (Lawler *et al.*, 1999). Altogether, it is estimated that the cost of erosion to the farming community is of the order £10 million yr^{-1} (Evans, 1995).

Figure 1.33.1 Areas of Britain most at risk of erosion (see text for details)

Although not important over the short-term, the loss in agricultural productivity over the longer term may be substantial. Evans (1996) has estimated a 10% loss of productivity of arable land since it was cleared of its woodland, equivalent to £700 million yr^{-1}, a figure which will slowly increase over time.

1.33.6.2 Off-farm

Off-farm impacts include sedimentation of reservoirs, pollution of watercourses by sediments, phosphorus and pesticides, which may travel attached to soil particles, and flooding by soil-laden runoff of property and roads. Flooding of property by runoff from agricultural land was noted by Morgan (1980) and the first detailed study was by Stammers and Boardman (1984). The South Downs, owing to the proximity of arable land to urban areas, became a focus of interest with 138 incidents of property damage recorded in the years 1976–2001 (Boardman *et al*., 2003a,b).

A five-fold increase in the area under maize in the UK since 1985 (MGA, undated) has has led to concern about the impact of eroded sediment on watercourses (Anon, 1996), and in particular the impact of the persistent pesticide atrazine (Alliston and Conway, 1995). Maize fields are at risk of erosion in both early summer and post-harvest. Areas where erosion and pollution have been recorded include the Yeo catchment in North Devon (Clark, 1999) and the Rother valley in West Sussex (Shepheard, 2003).

Off-site impacts of the pesticide aldrin have also been recorded by Harrod (1994). Pesticide applications to daffodil fields in Cornwall have been made for over 30 years. Frequent runoff events from low-intensity winter rain (2 mm h^{-1}) can pollute watercourses and damage fish stocks.

The costs of erosion off the farm are much greater, by an order of magnitude (Table 1.33.5). They include costs attributed to flooding and damage to property, repairs of footpaths and mending eroding stream banks and alleviating sedimentation of channels, improving fisheries and, most importantly, the costs of improving drinking water quality. It is likely that cost to repair footpaths is of the order of £1–2 million yr^{-1} for the whole of Britain (extrapolated from Evans, 1995), and the cost of alleviating sedimentation of rivers is of the order of £7 million yr^{-1} (Sear and Newson, 1991).

The cost of the dredging of rivers and estuaries to make them passable for ships is unknown, as are the costs of removing sediment from watercourses and reservoirs. Also, it is difficult to cost the loss of amenity or the aesthetic impacts of erosion (Evans, 1996) – eroding footpaths are unsightly. Of more serious concern, however, are those intangibles which cannot be costed (FHRC, 1983), such as stress and ill-health brought on by the expectation of flooding or the loss of items of sentimental value.

The estimated costs vary considerably (Table 1.33.5), especially those for damage to property by flooding. This variability may be partly attributable to when the estimates were made. Hence, the earlier (although later published) estimate of the cost of making water potable in the early 1990s (Evans, 1996) was much less than

TABLE 1.33.5 Some costs of erosion from arable land in England and Wales

Cost (£ million)	Source
Water pollution (pesticides, eutrophication)	
260	Evans (1995)
208	Pretty *et al.* (2001)
168	Environment Agency (2002)
Damage to property by flooding	
3	Evans (1995)
14	Pretty *et al.* (2001)
115	Environment Agency (2002)

the second estimate (Evans, 1995). The second was made when it became apparent how costly it was to install new water treatment works to cope with polluted water. The later estimate by the Environment Agency probably reflects running costs more than capital costs, as many treatment plants have now been uprated.

With regard to costs to right flood damage to property, there is a major discrepancy between the earlier estimates and that made by the Environment Agency. The latter organization has presumably factored in the costs of the autumn 2000 floods. However, the sequence of storms which produced the floods was a rare event (CEH, 2001). Also, the extent of erosion may not have been as widespread as was runoff. Thus, the storms saturated the ground but they were often of prolonged duration rather than intense, so that although sheetwash and runoff were widespread, rills and gullies were not (Evans' fieldwork). Erosion was further restricted in extent because much land was too wet to be cultivated for autumn-sown crops and so remained not ploughed but under protective stubble and weed until the following spring. Boardman (2003) notes that erosion was less extensive on the South Downs in autumn 2000 because the area under winter cereals was considerably reduced.

Whatever the reasons for the variability in costs, all estimates of the costs of erosion to the community are substantial. It is these costs which are driving the need to combat erosion.

1.33.7 SOIL CONSERVATION

Until the late 1990s, there was little institutional commitment to soil conservation in Britain, although pamphlets on water and wind erosion were published in the 1980s (MAFF, 1984, 1985). There was the perception among British farmers that erosion (both on- and off-site) was of little concern (see above). Rarely were they directly affected by erosion, for instance by gullying of their fields which interfered with agricultural operations or blows which removed seedlings (Evans, 1996). Until the late 1990s, farmers were hardly aware that the soil particles, nutrients and pesticides which washed off their land caused problems to the water supply industry. The impacts of sediment from eroding moorland and forest drains on reservoir water filtration plants in the uplands, and the incurred costs, were realized a decade or so earlier, resulting in guidelines to stop erosion (Forestry Commission, 1993). Where erosion had occurred in the arable lowlands, conventional approaches to conservation or runoff control such as minimum tillage, grassed waterways and buffer strips had little take-up in Britain.

More recently, European moves to reduce food surpluses, to cut agricultural subsidies and therefore, and *en passant,* to encourage land-use change, offered the opportunity to address environment issues. These changes were accompanied by changes in attitudes to soil erosion and conservation and can be seen in government and government agency publications (Boardman, 2002). In the late 1990s, a flurry of well-researched publications were introduced (MAFF, 1997, 1999a,b), plus the *Code of Good Agricultural Practice*, which discusses erosion and conservation (MAFF, 1998). However, a division of responsibility for erosion on the farm, and for off-site impacts, is reflected in the publication by the Department of the Environment of studies of erosion and flooding and environmental effects of agriculture (DOE, 1995a,b; DETR, 1998), although the latter was produced by an agricultural advisory agency. The recent creation of DEFRA (Department for Environment, Food and Rural Affairs) may help repair this unnatural separation. The Environment Agency is also showing interest in erosion as a source of pollutants to watercourses. This has led to advice to farmers as to how to avoid erosion and runoff (Environment Agency, 2001). Schemes such as the Rother Valley Landcare Project are driven by the need to reduce sediment and agricultural chemical pollution of a valuable fishing resource, the trout fishery of the River Rother. For this reason, the project is supported by the Environment Agency. High-value crops grown on easily worked and highly erodible soils (Fyfield 1 and 2, Shirrell Heath 1 and Frilford soil associations) mean that it is difficult to offer farmers sufficient incentives not to pollute. This relationship between workability and erodibility is a constant challenge and barrier to sustainable farming.

In parts of Britain dominated by arable farming, it is clear that there is a relationship between area of bare ground and the risk of damaging muddy floods. Evans and Boardman (2003) show that the strategic placing of small areas of grass (e.g. through set-aside or environmentally sensitive area schemes) on vulnerable slopes, together with the construction of small dams, reduces the risk of flooding even in exceptionally wet years such as 2000–01.

In 1996, the Royal Commission on Environmental Pollution reported on 'Sustainable Use of Soil' (RCEP, 1996). The report contains a short discussion of erosion and its impacts. Some five years later, the final draft of the government's 'Soil Strategy' is still awaited, although an 'Action Plan' is in preparation.

In the uplands, the concern in the last 10 years has been to boost farmer income and support rural populations. Growing concern with ecological change, especially in National Parks (loss of heather and related species), and overgrazing have prompted a shift from headage-based payments for sheep to management agreements which include limitation on numbers. Lowering grazing intensities, for example in Hey Clough and adjacent slopes (Evans, 1990d, 2005) and nearby moorlands (Anderson and Radford, 1994) in the Peak District, has reduced erosion. The Peak District Moorland Project has shown that badly degraded peat moorland can be restored (Anderson *et al.*, undated).

Public bodies such as the National Farmers Union, the National Trust and the Environment Agency have become involved, for various reasons, in soil conservation. All these organizations need to work together to protect the environment, the Environment Agency as the statutory body to oversee the environment and the farming community because it is their actions which often promote erosion. However, farmers take their actions in response to the prevailing economic (especially), political and social factors. It is these latter factors which need to be tackled by policy makers (Boardman *et al.*, 2003b).

1.33.7.1 Legal Issues

There are no legislative or legal constraints on farmers in Britain who allow erosion to occur on their land. However, if soil-laden water damages the property of a neighbour, the laws of negligence or nuisance may be invoked. For a successful prosecution it must be shown that the farmer knew of the risk to his neighbour (Boardman, 1994). It is likely that this can only be demonstrated when similar damage has occurred at the same site on a previous occasion or if it was foreseeable that damage or nuisance could be committed. The costs and difficulties of prosecution have deterred many potential litigants but several cases have been settled out of court. These include a recent case of runoff from outdoor pig fields in Suffolk that damaged properties in a village (Environment Agency, 2002; Evans, 2004); legally binding agreements were imposed by the court on the pig farmer and his landowner. Repeated flooding of Breaky Bottom Vineyard by runoff from winter cereal fields in 1987 was settled out of court (Boardman, 1994), but occurred again in 2000–01 and is the subject of current legal action. Local authorities may also invoke the Highways Act (1981), which allows them to reclaim the costs of damage to public highways from landowners. This Act has been used imaginatively by Isle of Wight Council to insist that landowners prevent runoff from their fields reaching highways (Boardman, 1994).

1.33.8 CONCLUSIONS

Over the last 30 years, since the issue of erosion was first brought to public attention (Evans, 1971), the emphasis has shifted from on-farm to off-farm impacts (Boardman, 2002). This is partly because monitoring schemes have shown that in most parts of the country the rates of erosion are relatively low, although questions still remain concerning upland areas. However, damage to property by runoff from agricultural land and pollution of watercourses from similar sources are now a focus of concern.

Significant progress has been made in recent years in addressing these issues. Better advice is now available to farmers; the Environment Agency is aware of the pollution issue (Environment Agency, 2001); set-aside can be used to reduce the risk of flooding; and legal action may be pursued if negligence or nuisance can be proved.

Climate change predictions for the future of warmer, drier summers and warmer, wetter winters in southern Britain suggest some increase in erosion on winter cereals and the expanding area of maize (Boardman *et al.*, 1990).

Erosion is largely driven by political and economic incentives. The way in which the land is farmed, rather than any vagiaries of the weather, is over the long term a more significant influence on runoff and erosion. Thus, erosion in Britain is a function of agricultural and land-use policy.

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1.34

Ireland

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1.34.1 INTRODUCTION

'The Emerald Isle': this description of the island of Ireland is to be found in almost every popular guidebook. Over-used the epithet, may be, yet from it, a geomorphologist might make some broad-brush, but still useful, deductions regarding soil erosion in Ireland. (Note that, throughout this chapter, 'Ireland' and 'Irish' refer to 'the island of Ireland', except where either of the two political entities is explicitly named.)

Clearly, Irish vegetation is seen as being green and lush. This implies both an adequate rainfall, and – provided the vegetation is not removed – protection of the soil from rainfall and from runoff. Hence it can be deduced that as long as adequate vegetation cover is maintained, soil erosion by water is not likely to be a major issue in Ireland. Adequate rainfall also suggests wet soils, and hence minimal wind erosion.

This is indeed the case. On the grassy lowlands of Ireland, although runoff is plentiful (and in places contaminated with agricultural pollutants, giving rise to some locally severe water quality problems), soil erosion by water is sporadic and generally uncommon; wind erosion is rare. However, the potential for soil erosion by water is there, and unsuitable land management has in the past, and could in the future, result in locally problematic erosion on the agricultural lowlands of Ireland. On the pastoral Irish uplands, overgrazing – a familiar story elsewhere in Europe – has led in recent decades to a loss of vegetation cover which is sufficient to trigger fairly widespread erosion of peat; commercial peat extraction may also contribute to occasional but dramatic 'bog bursts'.

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1.34.2 PHYSICAL GEOGRAPHY

Ireland has had a relatively complex geological history (see, e.g., the website of the Geological Survey of Ireland, 2004). Carboniferous limestone underlies much of the lowland Irish Midlands (Figure 1.31.1), with glacial deposits overlying the limestone. Being clay-rich, these have a low erodibility, in addition to low permeability. The central lowland is largely bounded by mountains, including the Mourne Mountains to the north and the mountains of Wicklow to the south-east. This mountainous rim is composed of a variety of rock types and rises to over 900 m in places. In the north, lowlands encircle Lough Neagh.

The Irish climate is maritime, and is strongly influenced by the proximity of the warm North Atlantic Drift and – over much of Ireland – by prevailing winds from the south-west. This results in a mild climate with a relatively restricted annual temperature range: mean daily January temperatures are typically 4 °C and for July/August typically 16 °C. Also, because of the Atlantic influence, Irish rainfall has a pronounced west–east gradient: mean annual rainfall on the central Irish lowlands is typically in the region of 1200 mm, whereas parts of the western

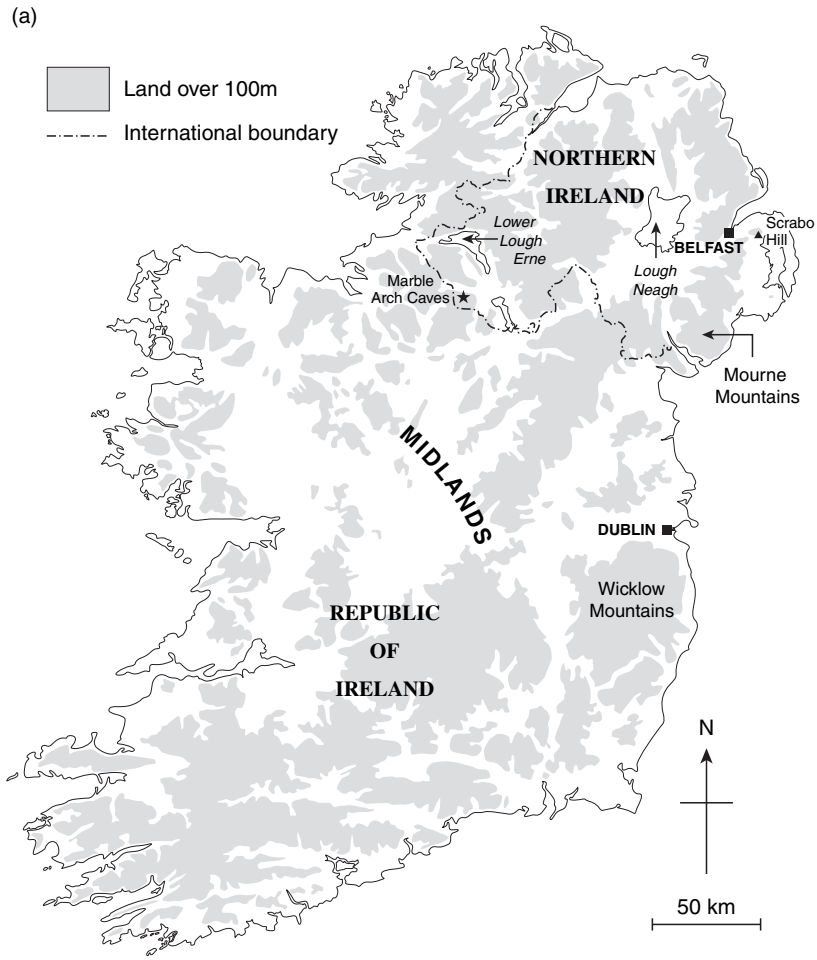

Figure 1.34.1 Ireland: selected physical features (a) and counties (b)

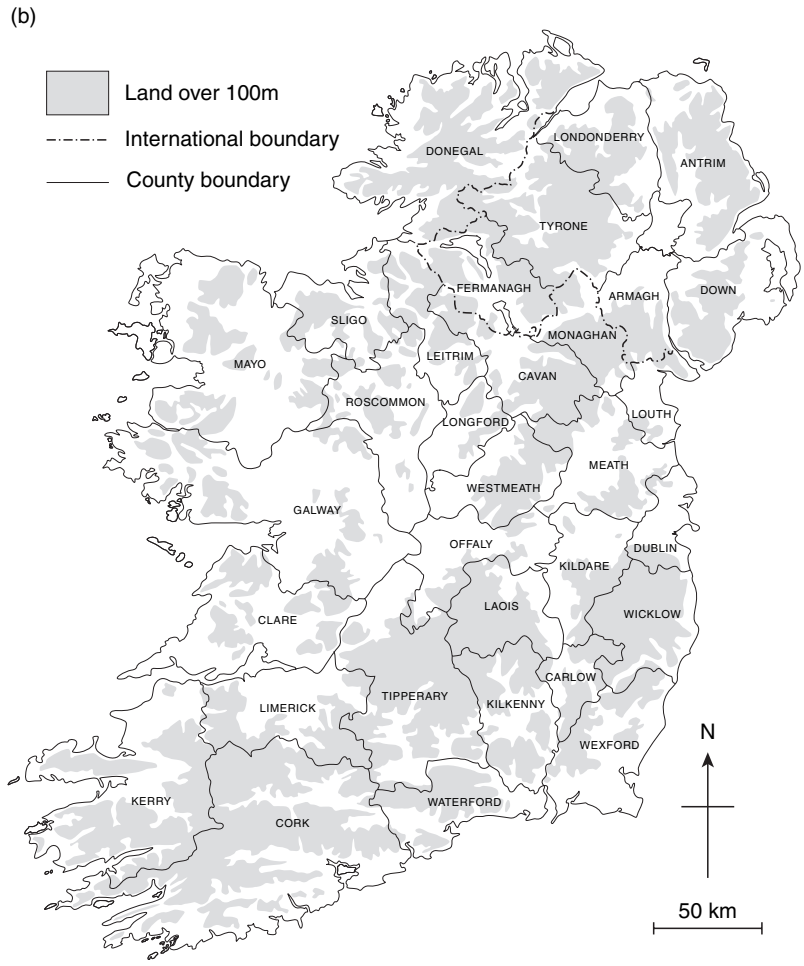

Figure 1.34.1 (*Continued*)

uplands receive over 2500 mm annually, with sheltered areas in the east of the island receiving only ca 750 mm. However, nowhere in Ireland does rainfall vary much seasonally. The wettest months are between August and January, with a rather more pronounced winter precipitation maximum on upland areas.

The equable Irish climate means that soils are not often subjected to extremes of drought or to freeze–thaw conditions. Leaching, gleisation and calcification are the principal pedological processes operating in Ireland, resulting in Podzols, brown and grey–brown podzolics, brown earths, Gleys, Rendzinas, Regosols, Lithosols and blanket and basin peats (Gardiner and Radford, 1980; Cruikshank, 1997).

The generally low permeability of Irish soils, coupled with stream gradients which are often very gentle in their lower courses, mean that parts of Ireland generate values for runoff per unit area which are among the highest in Western Europe (Wilcock, 1997). This has implications for the off-site transport of agricultural pollutants in runoff: nitrate and, more recently, phosphate from agricultural fertilizers have notably degraded the water quality in the major Northern Irish lakes of Lough Neagh and Lower Lough Erne (e.g. Anderson, 1997; Watson *et al.*, 2000, Watson and Foy, 2001).

The first period showing clear evidence for occupation throughout Ireland is the early Bronze Age, from about 4000 BP, although impacts of this early occupation are more apparent in upland regions. The Irish lowlands were previously largely covered by mixed woodland, in which there is often little evidence for settlement before the Early Christian period, i.e. from about 1500 BP (McCabe and Hirons, 1986); as agriculture developed, small-scale clearance gradually removed the trees. There are, however, indications of earlier settlement in a few lowland locations: for example, at a site in Co. Tyrone (Hirons, 1984), there appears to have been notable human activity from the Neolithic (ca 5500 BP), with forest clearance occurring later, in the early Bronze Age. In parts of Ireland (mainly the North), intensification of farming began early in the 17th century following immigration of English and Scots. By the 19th century, woodland covered only 1 % of Ireland's land area (Brogan and Crowe, 2003). Unsurprisingly, peat bogs now occupy much of upland Ireland.

At present, the Republic of Ireland has approximately 69 % of the land area devoted to agriculture (including common rough grazing), with a further 14 % classified as forestry or semi-natural areas. On this agricultural area, over 91 % is under grass, with 74 % being under permanent pasture or hay (Central Statistics Office, 2002). For Northern Ireland, around 80 % of the land area is now used for agriculture (including common rough grazing), with a further 6 % devoted to forestry. Around 95 % of the farmed area is under grass (Department of Agriculture and Rural Development, 2003c).

1.34.3 SOIL EROSION ON THE IRISH LOWLANDS

Despite the island's moderately high rainfall, which arrives more or less year-round, and the resulting high runoff, soil erosion by water is minimal at present on all lowland areas of Ireland. This is because of the almost omnipresent grass cover, and – to a lesser extent – because soils are clayey and hence have a rather low erodibility.

However, isolated instances of water erosion can occur where soils are unprotected, particularly where they are also more erodible. In Northern Ireland, there is occasional water erosion on areas of sandy soils, e.g. at Comber, south-east of Belfast, under potatoes (occasional minor wind erosion also occurs on the light soils here), on the slopes of Scrabo Hill in the north of the Province and along the south of the Mourne Mountains (Smith B, personal communication, 2004).

Soil erosion by water on the Irish lowlands was, however, more widespread in the past. Evidence for past erosion has been found in both geomorphological and archaeological studies. It includes colluvium accumulation along field boundaries (McEntee and Smith, 1993; McEntee, 1998) and depositional horizons in slope-bottom soil profiles (e.g. Culleton, 1975). For example, at the Co. Tyrone site mentioned previously, early Bronze Age forest clearance resulted in soil loss (Hirons, 1984), and evidence for Iron Age/Early Christian soil erosion has been found in the lowlands of southern Co. Down (Singh and Smith, 1973). Flax, the raw material from which linen is made, has a long history of cultivation in Ireland and is particularly associated with the 17th century intensification of agriculture. By the 18th century, it was widely grown in Northern Ireland (Hall, 1993). Spring-sown flax is known to be a crop which is prone to erosion by water (e.g. Souchère *et al.*, 2003), hence it is unsurprising to find lake-bottom sediments associated with former flax cultivation in Co. Down (Hall, 1990) and in Lough Neagh (Smith B, personal communication, 2004).

Off-site pollution of Irish water bodies from agricultural runoff is a serious present-day environmental issue in both the Republic of Ireland (e.g. Brogan and Crowe, 2003; Matthews, 2003) and in Northern Ireland (e.g. Wilcock, 1997; Withers *et al.*, 2001): Lough Neagh is considered to be one of the most eutrophic lakes in the world (Department of Agriculture and Rural Development, 2005a). Watson and Foy (2001) examined N and P budgets from grassland in Northern Ireland, and concluded that winter spreading of manures is a major causative factor for P pollution in particular. They note that a suitable hydrological connectivity between P source and watercourse is necessary for such pollution to occur; in Northern Ireland's clayey soils, an extensive system of under-field drainage provides this connectivity.

1.34.4 SOIL EROSION ON THE IRISH UPLANDS

A widespread problem on Irish upland areas is overgrazing by sheep. Erosion of peat on upland areas has been occurring for centuries (e.g. Huang, 2002) and, as in the lowlands, there is evidence of past erosion on the Irish uplands associated with the intensification of agriculture, from e.g. layers of gravel incorporated into peats and associated pollen evidence in the Mournes (Smith and Hirons, 1985). However, dramatic increases occurred in the size of sheep flocks in the 1980s owing to changes in the subsidization of agriculture in European Less-Favoured Areas. The resulting widespread decrease in vegetation cover leaves considerable areas of peaty soils increasingly exposed to rain, wind and freeze–thaw action; this leads to a removal of peat which has both on-site impacts (e.g. decreased habitat quality; reduced aesthetic appeal of afflicted landscapes) and off-site impacts (e.g. decreased water quality; disruption of downstream habitats, such as breeding grounds of Atlantic salmon). Up to 20 % of highland areas in the Republic of Ireland may be affected (Foss *et al.*, 2001): the worst-afflicted areas appear to be Galway and Mayo (Brogan and Crowe, 2003). Erosion of peat is also a notable problem in upland Northern Ireland (McGreal and Larmour, 1979; Tomlinson, 1982).

Occasional mass movements of peat ('bog bursts' or 'bog slides') have long been described by Irish historians (e.g. Praeger, 1897), with several taking place in recent decades in the uplands both of the Republic of Ireland (e.g. Alexander *et al.*, 1986; Coxon *et al.*, 1989) and of Northern Ireland (Colhoun *et al.*, 1965; Tomlinson, 1981). Although heavy rainfall is the immediate driver for these mass failures, in some cases it is probable that the commercial extraction of peat for fuel and horticultural purposes also plays an important role. Some work has been done on the impacts of peat extraction at Marble Arch Caves, near the boundary between the Republic of Ireland and Northern Ireland. Results from this study are, however, still unpublished (Gunn J, personal communication, 2004).

Water erosion also afflicts isolated locations on the uplands where vegetation cover is removed by hillwalkers, e.g. in Northern Ireland, along footpaths in the Mourne Mountains (Lowther and Smith, 1988; Ferris *et al.*, 1993) and in the Republic of Ireland, on the Wicklow uplands (Kane, 1998).

1.34.5 POLICY

Recent writers emphasize the need to implement policy which explicitly focuses on the conservation of Irish soils (Brogan *et al.*, 2002). To some extent, a greater awareness of soil conservation is being achieved by means of farmer-oriented handbooks produced both in the Republic of Ireland (Department of Agriculture, Food, and Rural Development, 2002) and in Northern Ireland (Department of Agriculture and Rural Development 2003b,c). Nonetheless, much remains to be done, including systematic monitoring of present-day erosion problems, and mapping areas which might be susceptible to erosion under changed land use. In the Republic of Ireland, progress has been made on a number of related objectives, including developing a set of indicators for soil quality and designing a national soil quality monitoring network (Brogan and Crowe, 2003). In Northern Ireland, a map of erosion-prone areas is in preparation (Jordan C, personal communication, 2004).

1.34.6 FUTURE SOIL EROSION IN IRELAND

Future anthropogenically driven climate change seems likely to bring to Ireland moderate warming, decreased summer rainfall and minor increases in winter rainfall (Betts, 2002a; Sweeney and Fealy, 2002; Environment and Heritage Service, 2004). Ongoing work also suggests an increase in the occurrence of high-intensity rainfall events in autumn (Crawford T, personal communication, 2005).

Such changes in Irish rainfall (particularly any increases in high-intensity events) may make bog bursts more common (Whalley and Favis-Mortlock, 2002). In the lowlands, any move by Irish farmers to take advantage of this warmer and possibly drier growing environment will probably mean a move away from near-ubiquitous year-round grass to arable crops. The resulting decrease in cover, particularly if it occurs during autumn, is likely to exacerbate pollution of runoff by agricultural chemicals (Betts, 2002b). It may also once again make soil erosion by water a problem in susceptible locations.

1.34.7 CONCLUSIONS

Soil erosion in Ireland, although a relatively minor problem overall, can nonetheless be of moderate local importance, particularly on overgrazed areas of the uplands and on small patches of sandy soils in the lowlands. Off-site pollution resulting from agricultural runoff is, however, a serious issue locally.

This generally fortunate position is, despite year-round rainfall, due to a near-universal grass cover. Complacency must, however, be avoided (particularly in the lowlands), since any future shift to arable farming, possibly driven by climate change, would be likely to reduce soil protection. Soil erosion has been a problem in Ireland in the past: it could again become one in the future.

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Section 2

Introduction

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2.1

Past Soil Erosion in Europe

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2.1.1 INTRODUCTION: THE IMPORTANCE OF A HISTORICAL CONTEXT FOR SOIL EROSION RESEARCH

Those who cannot remember the past are condemned to repeat it

George Santayana, *The Life of Reason*, Volume 1, 1905

When we look at the present-day soils in many European landscapes, it is immediately obvious that Santayana's statement is also important in terms of soil erosion: during the centuries phases of landscape stability and soil formation were followed by phases of land use and soil erosion – in some cases even soil degradation – until agriculture ceased and another phase of soil formation began. Thus today, erosional forms and sediments related to past land use can be found widespread, such as deeply truncated soils on slopes, ancient rills and gullies, and even plough marks, colluvial deposits on the lower slopes and clastic alluvial deposits in the floodplains. As a result, the evolution and distribution of contemporary soils can only be understood by taking into account impacts of the past.

The past is the key to the present and the future

IGBP PAGES

The palaeo-perspective is an essential research focus in much of global change research. Dearing (2002, IGBP PAGES focus 5: HITE) lays out the general research agenda, which can easily be adapted for soil

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erosion. [PAGES: Past Global Changes project of the International Geosphere – Biosphere Programme (IGBP) established by the International Council for Science (ICSU). Human Impact on Terrestrial Ecosystems (HITE) activity in the thematic focus 5 of IGBP PAGES.] Thus, reconstructing past soil erosion is necessary to:

- provide long-term trajectories of soil and landscape change up to the present;
- unravel background or pre-human impact conditions (e.g. 'natural' erosion rates) with which modern rates can be compared and judged;
- quantify natural variability of erosion processes and define threshold conditions for change (e.g. extend the temporal coverage of observations);
- provide historical analogues for extreme events, abrupt impacts and human–environment interactions;
- evaluate the relative impacts of climate and human activities on processes through time;
- develop and test predictive models by providing time series and system dynamics at appropriate temporal and spatial scales.

Especially in Europe, with its long and diverse history of land use, the impacts of past soil erosion are manifold and the historical perspective (*historical* in the sense of *past* and relating to the period before process measurements and not *sensu stricto* the period of written documents only) should be an integral part of any research that tries to understand soil–landscape systems. This is especially obvious where soil erosion in the past was severe and original soils were only shallow. In several landscapes, such as the South Downs in England (Favis-Mortlock *et al.*, 1997) and the limestone hill country in southern Germany (Lang *et al.*, 2003a), soil erosion was sufficient to remove almost completely Holocene soils and Pleistocene sediments (mainly loess) already before the Iron Age. This has clear implications for the present day:

- The present day soils do not represent the Holocene climax soils;
- The present agricultural use is constrained (often limited) owing to impacts of past land use.

Reconstructing soil erosion of the past is not an easy task. Here we discuss scientific approaches and techniques that are specific for palaeo-studies and present some results from characteristic case studies from north-western, central and south-eastern Europe. We show that understanding the present soil landscape in Europe is only possible by taking into account the longer term history of soil erosion and show that system functioning itself is strongly contingent on the history of change.

2.1.2 SCIENTIFIC APPROACHES AND METHODS

The methods used to quantify past erosion differ clearly from the methods used for studies of present-day processes and are much more related to methods that are used for Quaternary studies. Instead of direct process measurements, palaeo-studies have to rely on the preserved sedimentary and morphological records. Instead of using high-precision electronic clocks, they have to rely on chronometric techniques or historical sources. Instead of being able to constrain experimental conditions (limit catchment size, isolate process domains, trap sediment), in palaeo-studies one often needs to analyse the full range of possibilities. Thus, on expanding the time-scale the complexity of the system increases (overview in Phillips, 2003). The immediately obvious consequence is that the precision of results from palaeo-studies must be substantially reduced compared with those of process measurements.

The major challenges that must be faced when targeting the past are:

- The state of a variable (dependent or independent) is time reliant. In many European landscapes, soil erosion factors such as slope length, slope gradient and soil erodibility – which are usually set to be constant in contemporary process studies – change over centuries and millennia.
- Sediments usually do not represent complete records. With the exception of lake sediments, deposits that are stored in most continental sedimentary sinks will not all be preserved over time but might be subject to later erosion.
- The coupling of slopes and sedimentary archives changes through time (Dearing and Jones, 2003). As long as there is, for example, sufficient accommodation space on the lower slope, it will act as an efficient sedimentary sink. When, after some decades of soil erosion, this trap is filled up, a higher percentage of the sediment eroded on the slope will be transferred across the lower slope and into the rivers (Lang and Hönscheidt, 1999).
- Thresholds in system behaviour will change through time. Similar inputs can lead to dramatically different responses depending on the evolution of a system's sensitivity (Brunsden and Thornes, 1979; Schumm, 1991). Agricultural landscapes are more sensitive to climatic variability than natural landscapes because tillage and grazing typically reduce water infiltration and increase rates and magnitudes of surface runoff (Knox, 2001).
- Different processes can lead to similar deposits (equifinality). In many cases it will not be possible to differentiate if – for example – a soil erosion-derived colluvium was formed in response to sheet and rill erosion or gullying (Nemec and Kazancy, 1999).

Comparing soil profiles at eroded sites with profiles at preserved sites allows the reconstruction of total soil truncation since agriculture started. Unfortunately, no information about the timing and intensity of past erosion can be obtained. There are techniques available to determine erosion rates at a spot (e.g. based on *in situ*-produced cosmogenic isotopes), but these usually are only applicable for much longer time-scales ($>10^5$ yr). Points in time can often be reconstructed from archaeological finds. Prehistoric structures often allow the reconstruction of land surfaces contemporaneous with the prehistoric remains (Lang *et al.*, 1999). However, again such information is temporally discontinuous. Temporally resolved information therefore has to rely on translocated soil particles that are trapped in sedimentary deposits. Different types of sedimentary archives originate from soil erosion and can be used for reconstructions: (1) slope deposits, (2) alluvial sediments, (3) lake sediments and (4) coastal and marine sediments.

1. Slope deposits accumulated on the lower slopes or in gully fills can be used to derive detailed information on past erosion on the adjacent hillslope (e.g. Lang and Hönscheidt, 1999; Bork *et al.*, 2003). These sediments are rather difficult to analyse but potentially offer the highest resolution as the majority of eroded soil is stored already on the slopes.
2. Alluvial sediments have been used extensively to gather information on smaller and larger catchments (e.g. Macklin, 1999). Analytical techniques for this type of sediment are well developed. Results usually cannot be linked to specific slopes but to catchments. Internal dynamics of rivers have to be well understood in order to extract the soil erosion signal from alluvial sediments (e.g. Trimble, 1999).
3. Lake sediments form complete records and are used to extract spatially averaged but temporally highly resolved erosion rates. Many lake deposits offer the possibility of looking at annual resolution (e.g. Foster *et al.*, 1985, 1990; Dearing and Foster, 1987; Dearing *et al.*, 1990, Zolitschka *et al.*, 2000).
4. Coastal and marine sediments: translocated soils were transported into the coastal areas but extracting soil erosion records from coastal deposits is difficult. Several studies have shown enhanced sediment influx following human impact during the Holocene (overview in Long, 2001). Owing to climatic and sea-level

fluctuations, the cause of this enhancement has so far rarely been determined unambiguously. Only recently have marine sediments successfully been used: Oldfield *et al.* (2003) reconstruct human activity in the sediment source area from deposits taken from the central Adriatic Sea.

Colluvial and alluvial deposits have the advantage that they are coupled more closely to the erosion events on the slopes and can therefore be used to determine spatial variations with high resolution. Detailed information was successfully derived for small catchments (e.g. Dotterweich, 2003; Schmidtchen and Bork, 2003). As sediments move from source to sink, the lag times due to intermediate storage (up to centuries and millennia), fluvial erosion, sedimentation and reworking make reconstruction of a soil erosion history on the slopes more difficult. Several authors have addressed the problems associated with sediment propagation (e.g. Walling 1983, 1987; Wasson, 1996; Syvitski, 2003). Before being deposited, eroded soil particles are transported over a certain distance depending on landform, vegetation and magnitude of a runoff event: Particles transported by tillage will be deposited within a field parcel. Particles transported by water during low- and medium-magnitude rainfall events are mainly deposited at concave lower slopes, in shallow zero-order basins and in small alluvial fans. During high-magnitude rainfall events, different processes are operating and eroded particles are often evacuated from the slopes and transported to the rivers (Lang *et al.*, 2003b).

Owing to these problems, integrated studies are needed that take into account all types of storage and try to derive more robust and quantitative results. These studies are usually based on constructing sediment budgets for different periods. In addition to its use in geological research, this approach has proved to be very helpful for the study of past soil erosion on the shorter time-scales of decades (e.g. Trimble, 1999). At the moment, studies of the full Holocene history of soil erosion with some temporal resolution are still rare (Macaire *et al.*, 2002; Foster *et al.*, 2003). At present the most promising approach for more quantitative results is to combine mathematical modelling with information extracted from sedimentary records (Lang *et al.*, 2003c; Preston and Schmidt, 2003). If independent records of climate and land-use history are available to drive the models, it should be possible to construct more complete pictures of a region's erosion history (Lang *et al.*, 2003b). Records of past temperature and humidity have recently become available from historical sources (e.g. Glaser, 2001) and ice cores and sedimentary archives (e.g. Alverson *et al.*, 2003). Detailed records of land-use patterns and farming techniques based on historical sources (e.g. Burggraaff, 1992) or archaeological information (e.g. Lüning, 1997) are also available, but their spatial coverage is still very limited.

Further details on the techniques and methods that are applied to reconstruct past soil erosion from sediments can be found in Bork and Lang (2003).

2.1.3 EUROPE'S SOIL EROSION HISTORY

Past soil erosion is as widespread as past land use and is as old as the first farming activities during the Neolithic period. In Europe, with its long and diverse history of land use, the great majority of present-day soils have somehow been transformed by human impact. However, the transformation history varies widely for different regions. This is partly due to the different natural settings, which is nicely documented in this volume. However, different from today, levels of land-use technology were not similar at a given time. Especially in the early periods, the spread of technology took a long time, starting in the eastern Mediterranean and not arriving in north-west Europe until 2000 years or so later.

The widespread occurrence of past soil erosion is reflected in the wealth of case studies that exist from all over Europe. The great majority of information on past soil erosion was gathered in the framework of genetic studies and using more deductive approaches. Especially the sub-discipline of geoarchaeology has contributed significantly to our present understanding of the effects and causes of past erosion. Over the years,

explanations of past soil erosion evolved rather dramatically: for example, in the Mediterranean (reviewed in Bintliff, 2002) initially monocausal reasoning was used and first 'natural' (= climatic) processes (e.g. Vita-Finzi, 1969) were thought to be responsible for past soil erosion. Later, in a similar monocausal fashion, 'anthropogenic' processes only were claimed to cause soil erosion (e.g. Van Andel *et al.*, 1986). At present, more pluralistic multicausal reasoning is applied. However, still today, the great majority of studies derive rather descriptive information and quantitative results are largely missing. This is partly due to the difficulties inherent to palaeo-studies (as above), but also to the different scientific approach used by the more classical genetic studies. In addition, the complexity of land-use response to climatic change (e.g. Berglund, 2003) adds several degrees of freedom to the way in which palaeodata can be interpreted.

Several authors have produced reviews on past erosion studies: Bell and Boardman (1992) and Dearing (1994) give excellent synopses. A recent compilation of case studies from fluvial sediments was put together by Howard *et al*. (2003) and from lake sediments by Brauer and Guilizzoni (2004). Regional overviews of impacts of past erosion can be found for Central Europe by Bork *et al.* (2003), Kalis *et al* (2003) and Zolitschka *et al*. (2003), for Poland by Klimek (2002, 2003), for the Mediterranean by Grove (1996), for nothern Italy by Marchetti (2002), for France by Neboit-Guilhot (1991) and Macaire *et al*., (2002), for Belgium by Verstraeten *et al*. (see Chapter 1.30) and for Great Britain by Macklin (1999) and Edwards and Whittington (2001).

Here we will review some recent studies from three contrasting areas: south-eastern Europe with almost 9000 years of soil erosion, central Europe with almost 7000 years and north-western Europe, where the history of human impact is rather short but nevertheless dramatic.

2.1.3.1 South-eastern Europe

Today in many parts of south-eastern Europe, soils are almost missing and unweathered rocks and sediments are exposed at the earth surface. On the slopes, remnants of Holocene red and brown Mediterranean soils (Yassoglou *et al*., 1997) are found mainly in erosion-protected depressions or buried under colluvium on the foot slopes. This clearly indicates severe soil erosion. Clearly, modern agricultural practices contributed substantially to the overall loss of soil, but past processes also had severe impacts.

Fuchs *et al*. (2004) showed from soil erosion-derived colluvium that on the Greek Peloponnesus peninsula colluvium formation was dominated by the intensity of land use. Holocene climatic fluctuations seem to be of only secondary importance, as sufficiently erosive rainfall events may have occurred during all agricultural periods. During the early Holocene, before agriculture started, Fuchs *et al*. (2004) found very low sedimentation rates. With the onset of the Neolithic in the 7th millennium BC, sedimentation rates increased, stay high during the Neolithic and decreased in the following Chalcolithic and Early Bronze Age periods (4500–2050 BC). Higher rates are found for the Middle Bronze Age to Early Iron Age and the beginning of Classical Antiquity. Very high sedimentation rates occurred at the end of Classical Antiquity and during the Roman period. The sedimentation rate decreased during medieval times and since then it has increased again.

The general conclusion of Fuchs *et al*. (2004) is that the pattern of sedimentation matches the pattern of cultural development and population density. The critical factor for soil erosion was the sensitivity of the land surface to erosion, and thus the size of the arable land and the intensity of agricultural practices. Phases of high colluviation coincide with known periods of higher settlement density and pronounced farming activities. Rates of reduced colluviation occurred during periods where also the settlement density was reduced. The exception is the Early Bronze Age, where the settlement activity was high but low sedimentation rates were detected. According to the authors, this may be explained by the introduction of soil conservation measures (probably terracing).

From the basin of Drama (eastern Macedonia, Greece), Lespez (2003) reports distinct phases of soil erosion and stream aggradation over the past 7000 years, and ties them directly to long-term land-use changes. Alluvial fill

accumulated rapidly during the Middle and Late Holocene. For the Late Neolithic to Early Bronze Age (5400–2000 BC), low levels of aggradation were detected. Moderate rates of alluviation occurred during the Late Bronze Age (1600–1000 BC), high rates in the Antique and the Early Byzantine Era (3rd century BC–7th century AD) and the highest rates in the Ottoman period (15th to early 20th century AD). Lespez (2003) explains the late onset of aggradation – almost three millennia after the first farming activity and the onset of erosion – by the settlement pattern during the Neolithic and Early Bronze Age periods. Early farmers preferred to cultivate more stable soils on gentle slopes. Only when during the Late Bronze Age the land-use pattern changed and less stable soils and steeper slopes were cultivated did alluvial aggradation increase. According to Lespez (2003), the two periods of accelerated alluviation in historical times are also mainly linked to land-use changes: deforestation and the extension of agriculture into more sensitive mountainous areas. These two impacts also enhanced the sensitivity of the river system: during the Ottoman period, already modest changes in climate led to strong aggradation.

The two studies show that in south-eastern Europe soil erosion occurred already during the Neolithic – 9000 years ago.

2.1.3.2 Central Europe

Especially in the loess-covered areas of central Europe, impacts of past soil erosion are widespread and dramatic. The majority of soils are strongly truncated and often soils are completely missing. A large body of results from case studies on impacts of former agriculture is available from this region, but attempts to regionalize results are rare. Here we portray initial attempts from Bork and Lang (2003) to derive more regional pictures for Germany for the periods (1) from the Middle Ages to modern times and (2) for pre-1200 AD.

2.1.3.2.1 Middle Ages and Modern Times in Germany

Information on past soil erosion from more than 2200 study sites in south-east Lower Saxony was compiled (Bork, 1983, 1988) and integrated by clustering results for regions with largely similar substratum, soil evolution and soil translocation history. Then, in a hierarchical approach, a few km^2 large landscape elements were chosen randomly and within each landscape element catenas were selected, again randomly. Subsequently, for each catena the volume of eroded soil and the volume of sediment stored were calculated (Bork *et al.*, 1998). Finally, mean values were calculated for all landscape elements and taken to represent the whole region. Results show that at the upper and middle slopes an average of 2.3 m of soil is missing. More than 80% of this material was not transported out of the catchment but deposited on the lower slopes. At many study sites, high-resolution chronologies are available, allowing the determination of erosion volumes for specific land-use periods. This allowed the identification and quantification of single extreme events, e.g. in the first half of the 14th century (Bork, 1988; Bork *et al.*, 1998).

Data for other regions in central Europe are based on a different approach: for eastern Brandenburg, for example, catenas were selected that are typical of a larger area. From medieval to modern times, a mean soil loss of 0.5 m was determined, the main part of which occurred in the first half of the 14th century. This approach was also applied to other German landscapes. Finally, spatial averages for soil erosion were calculated for each landscape region. Areas with clearly different character, e.g. the Alps, were excluded from the analysis.

The results of this regionalisation attempt are given in Figure 2.1.1. Changes in land cover/land use and estimated rates of soil erosion are plotted versus time for the period since the Early Middle Ages. The high resolution of the data allows the correlation of soil erosion maxima with high-magnitude rainfall events. Extreme soil erosion occurred during the first half of the 14th century – a period during which extreme rainfalls coincided with the all-time low in woodland cover in Germany (Bork *et al.*, 1998). A second, less pronounced, extreme period is evident during the second half of the 18th century. This again is a period for which documentary evidence of extreme rainfalls and high runoff exists.

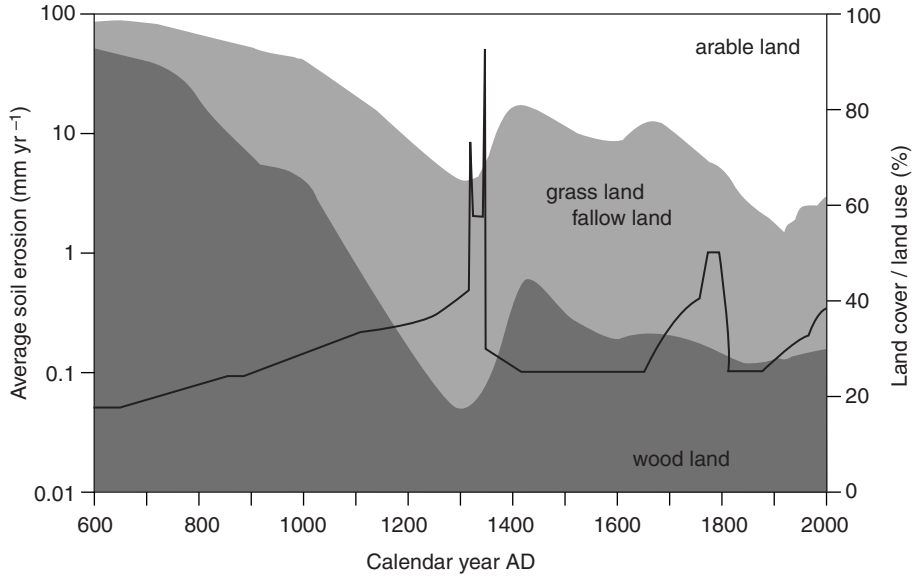

Figure 2.1.1 Land cover/land use (shaded) and soil erosion (black line) in Germany (excluding the Alps) since the Early Middle Ages (data from Bork *et al.*, 1998). The average soil erosion in mm yr^{-1} is plotted as solid line (left scale). For three land-use classes the proportion of land cover is plotted as grey tints (right scale). [Reproduced from Bork Hr, Lang A, Quantification of past soil erosion and land use/land cover changes in Germany. In *Long Term Hillslope and Fluvial System Modelling – Concepts and Case Studies from The Rhine River Catchment*, Lang A, Hennrich K, Dikau R (eds). Lecture Notes in Earth Science, 101. Springer, Heidelberg, 2003; 232–239, with permission from HR Bork]

2.1.3.2.2 South Germany Before the Middle Ages

For periods earlier than the Middle Ages – as usual when going further back in time – the quantity and quality of information are even further reduced. Written records are largely missing for Germany and, even where they are available, they should be treated with caution. However, approaches based on soils and sediments have to face more challenges: the sedimentary record is less and less complete as earlier periods are considered. The chance of older sediments being eroded is higher, as is the risk of a total overprint of traces of earlier soil formations. Also, chronometric information is harder to obtain as reworking of artefacts and organic remains is frequent and indirect dating approaches can be misleading (Lang and Hönscheidt, 1999).

Still, numerous local studies have been carried out in the loess hills of south Germany and therefore detailed information on soil erosion and sediment storage exists especially from the surroundings of archaeological sites. Unfortunately, for most of the study sites only stratigraphic and chronological information exists and volumes of erosion and deposition were not determined. Hence, the extrapolation of findings from local case studies to a more regional scale is problematic and at present only a first graphical analysis is available (Lang, 2003). OSL (optically stimulated luminescence) ages of soil erosion derived colluvial sediments were analysed to construct a frequency analysis of phases of soil erosion. The period covered is from the beginning of agriculture until 1200 AD. The frequency distribution was constructed by: (1) representing the OSL ages by Gaussian distributions and (2) summing all the single curves (Figure 2.1.2). A first significant increase in colluviation occurred during the Bronze Age. During the Iron Age/Roman period and at around 800 AD, distinct maxima appear in the distribution and the highest frequencies are present towards the end of the period analysed, around 1100 AD.

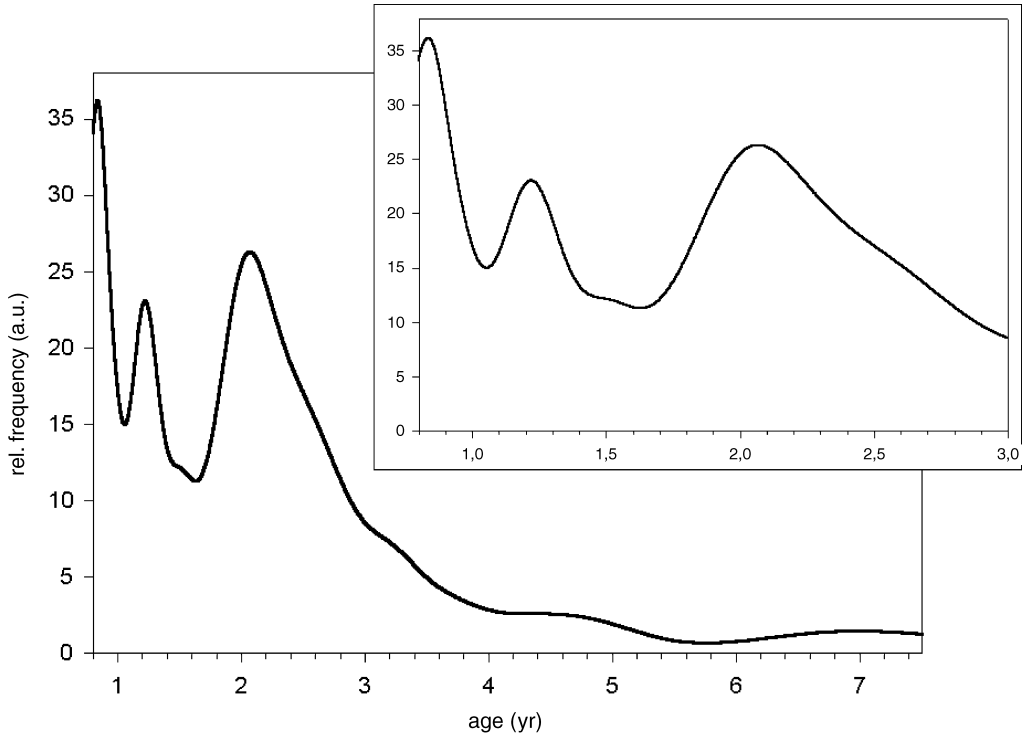

Figure 2.1.2 Probability density distribution of 60 OSL ages for soil erosion-derived colluvium from southern Germany for the period 0.8 to 7.5 yr. Inset: enlargement for the period 0.8 to 3.5 yr. (Reprinted from Lang A, Phase of soil erosion-caused colluviation in the loess hills of south Germany. *Catena* **51**: 209–221. Copyright 2003, with permission of Elsevier)

Conclusions that can be drawn from such an approach are restricted by the still rather limited amount of data, sampling bias and other factors. The oldest colluvial sediments were deposited during Neolithic times. Colluviation occurred more frequently during phases of stronger human impact such as the Iron Age and Roman periods, while the maximum number of optical ages relate to medieval times. This indicates that colluviation during this period was dominated by the intensity of land use. Climatic fluctuations seem to play a secondary role, considering that sufficiently erosive rainfall events occurred during all agricultural periods. Probably the critical factor was the landscape's sensitivity to erosion.

2.1.3.3 North-western Europe

North-western Europe has the shortest history of land use, but past impacts are widespread and responsible for many present soil characteristics (overview in Bell, 1992). Favis-Mortlock *et al.* (1997) simulated the effects of past erosion on a hillslope in the UK South Downs from 5000 BC to the present. According to their finding, the major period of soil loss was between 2000 BC and 200 AD, followed the permanent clearance of woodlands and the gradually intensifying agriculture. Already before the medieval period the Pleistocene loess and Holocene soils were stripped off the slope completely. Studies on valley fills also revealed the long-term impact of agriculture on north-western European landscapes: Wilkinson (2003) investigated colluvial sequences infilling dry valleys of chalk escarpments in southern England. He shows that different spatial and temporal patterns of sedimentation are due to regional variations in past land use, storm impact and

topography. Foster *et al.* (2000) identified medieval soil erosion from a minerogenic sediment deposit in a wetland in southern England. They link the origin of the deposit to increased soil erosion due to a series of wet winters in between 1200 and 1400 AD.

Edwards and Whittington (2001) reviewed results from 50 lakes on the British islands and analysed indicators for erosion. Accelerated erosion occurs only at lakes with clear indications of human impact. Lakes with uniform sedimentation through time are mainly located in northern Scotland and have no, very little or only rather recent signs of human impact. The beginning of the increased sediment accumulation usually occurs only after the first signs of human impact, thus showing a delay in system response. Ages for the beginning of increased sedimentation cluster at 3300–3000 BC 2500–2200 BC and 1000–800 BC, and broadly coincide with the early Neolithic, the mid-Neolithic and the Late Bronze Age, respectively.

Towards the northern, more marginal agricultural areas, past soil erosion had dramatic effects. Often the high sensitivity of the landscapes was paired with high vulnerability of the pioneer settlements. In detailed results from Iceland, Simpson *et al.* (2001) explain settlement success and failure by the presence or lack of appropriate grazing regulations and associated presence or lack of land degradation. In southern Iceland, where the period of occupation started at 874 AD, regulations to prevent overgrazing were in place already from ca 1200 AD onwards. For north-eastern Iceland, Simpson *et al.* (2004) show how adaptive land management techniques reduced erosion rates already in the 15th century below the regional average. Both studies prove that management practices were a major factor in past land degradation and important for explaining settlement success and failure, especially in agriculturally marginal regions.

2.1.4 CONCLUSIONS

The results presented here reflect only a limited extract from the large body of information available on past soil erosion in Europe and its significance for the present. Still, we hope that we were able to show that soil erosion is not just a modern problem. Past soil erosion was as widespread as past land use and is as old as the first farming in the Stone Age. Differences in the temporal pattern of erosion history across Europe reflect not only differences in natural settings but also the time lags in technological spread.

Amounts of soil erosion varied largely through time. Over the longer term, changing landscape sensitivity seems to be more important than climatic changes. Of course, extreme events leave their imprints in the landscape. However, the rainfall threshold for initiating soil erosion is much lower on arable land than under woodland. This sensitivity is determined by the type and intensity of a landscape's agricultural use. Especially for highly vulnerable pioneering settlements (Messerli *et al.*, 2000), landscape sensitivity was of immense importance. Without successful management practices, soil erosion and land degradation lead to settlement failure. This has been speculated to be a reason for the short lifetime (a few generations of inhabitants) of many Neolithic settlements in the loess areas of central Europe (after soil erosion had stripped off the uppermost soil horizons, farming techniques could not cope with the clay-enriched B-horizons that were then at the surface) and is clearly documented for a very different time and region: the medieval settlements in Iceland.

Especially during the Iron Age or medieval period, amounts of soil erosion were in excess of today's erosion in several areas. Many of the barren landscapes in the Mediterranean, but also in the hill landscapes of central and north-western Europe, are products of past soil erosion. Holocene soils were completely truncated already several centuries ago and the resultant rock surfaces are often without soil and without agricultural use today. Almost everywhere in European agricultural landscapes, pre-modern soil erosion significantly truncated soils. The present day soils can therefore only be understood by taking into account their erosion history.

The present state of knowledge is based on mainly qualitative and conceptual information. The further integration of new results will, it is hoped, allow refining of the history of soil erosion in Europe. Clearly, more quantitative results are needed. Most promising in this respect are approaches based on mathematical

modelling of soil erosion processes over the long term. Independent records of climate and land-use history are recently becoming detailed enough to be used as drivers for the models. The information extracted from sedimentary records could then be used to validate and calibrate the modelled scenarios. This should allow the integration of results from different scales and the construction of more complete and quantitative pictures of a region's erosion history.

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Soil Erosion Processes

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2.2

Soil Erosion in Europe: Major Processes, Causes and Consequences

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2.2.1 INTRODUCTION

Soil erosion is the detachment, entrainment and transport (and deposition) of soil particles caused by one or more natural or anthropogenic erosive forces (rain, runoff, wind, gravity, tillage, land levelling and crop harvesting). Large spatial and temporal variations in soil erosion processes and rates are observed in the European countries (see country chapters). The objective of this chapter is to explore the various soil erosion processes at the European scale, to analyse the major causes and consequences and to pinpoint major research needs.

Why is there a need for understanding soil erosion processes, their rates, extent and controlling factors at the European scale? Throughout Europe there is a large diversity of landscapes and of land use which causes significant variations in soil erosion processes and rates. Environmental management requires a thorough understanding of erosion process combinations in a given European environment. In general, we have a fair understanding of mechanisms of soil erosion and controlling factors. However, applying this knowledge to a given local context seems to be difficult. Hence there is still a need for research targeted at soil erosion-related topics such as processes, data on rates and factors, models, consequences including both on-site and off-farm impacts, control and soil conservation measures and strategies.

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2.2.2 FUNCTIONS OF SOILS AND THE THREAT OF SOIL EROSION

Soil erosion may affect soil functions to various degrees. Functions of soils determining soil quality can be summarised as follows: (a) food and fibre production function, (b) water filter function (c) ecological function (soil is the habitat for many micro-organisms, it maintains a genetic diversity of micro-organisms, stores nutrients and is the environment where roots grow), (d) bearing or foundation engineering function, (e) archive function (soils store artefacts and are testimony to past cultural history, land use or climatic change) and (f) heritage function (integrated into human culture; soils are important abiotic elements of landscapes which need to be conserved for future generations).

Most importantly, soils are the medium in which crops are grown. Without soils in good health crop yields will decline. Soil erosion leads to soil surface lowering and hence a reduction in soil thickness. If soil thickness decline is not compensated by soil formation, soil erosion threatens sustainable crop production. There is a close relationship between soil thickness and crop yield (Evans, 1981; Bakker *et al*., 2004). Loss of nutrients, along with erosion, affects crop yields. In extreme cases, soil erosion may affect yields owing to loss of seedlings and inability to harvest crops due to the presence of gullies.

The soil cover fulfils an important hydrological function. Under natural vegetative cover of woodland or grassland, soils have high infiltration rates (e.g. $>50\ \mathrm{mm\ h^{-1}}$) and high resistance to water erosion. Thus even under extreme rainfall conditions runoff is unusual and, if it occurs, clean (lacking sediment). The impact on flood events is therefore limited. All conservation and flood protection strategies recognise this relationship: well-vegetated ground encourages infiltration and limits erosion. Many strategies attempt to reduce the total amount of bare ground through the year or the length of slope that is bare at any given time.

Soil erosion has been recognised to have consequences both on- and off-site. If soil is lost or its quality is decreased then it is likely that many of its functions will degrade. This may happen over the short term through catastrophic loss, but it may also occur through long-term change, for example, reflected in a gradual change of hydrological response and hence a change in flood frequency. Off-site impacts of soil erosion are now recognised to be important in Europe particularly muddy flooding and damage to property (Chapter 2.19), sedimentation of artificial reservoirs (Chapter 2.20), eutrophication (Chapter 2.21) and damage to fish stocks.

2.2.3 THE PHYSICAL AND SHIFTING HUMAN GEOGRAPHY OF EUROPE AS A BASIS TO UNDERSTANDING MAJOR SOIL EROSION PROCESSES AND CONTROLLING FACTORS

Europe has important climatic, topographic/geomorphic, geologic/pedologic, land use and political gradients (see, e.g., Koster, 2005) affecting the type and rates of soil erosion processes, e.g. snowmelt erosion in Scandinavian countries and badland development (caused by water erosion and mass movement) in the Mediterranean. Northern, western and eastern Europe are characterised by the growing of cereals and root and tuber crops (e.g. sugar beet, potatoes), which affect water erosion (Chapter 2.4 and 2.5), tillage erosion (Chapter 2.9) and soil erosion during crop harvesting (Chapter 2.10).

Climate has a strong influence on soil erosion. Rain properties control the eroding capacity of the rain (= rain erosivity) and hence rates of soil degradation processes such as surface sealing and crusting (Chapter 2.3), interrill and rill erosion (Chapter 2.4), gully erosion (Chapter 2.5), pipe and tunnel erosion (Chapter 2.6) and landsliding (Chapter 2.8). Wind velocity determines wind erosivity (Chapter 2.7) whereas air temperature controls the occurrence of frost, snowfall, snowmelt and soil moisture, the last of which affects the susceptibility of soils to erosion (= soil erodibility; Chapter 2.15). It used to be thought that the 'light rains' of north-western Europe meant that there was no water erosion risk (Hudson, 1967). In the last 30 years it has become clear that the large quantity of rain, rather than the high intensity, falling on bare arable ground

can lead to soil crusting and/or saturation, runoff and erosion. However, intense summer thunderstorms may also play their part (e.g. Chapter 2.14). In the Mediterranean basin, seasonality of rainfall (and therefore vegetation growth; see Chapter 2.14) plus high-intensity storms have given rise to a long history of erosion and flooding. The highest recorded rainfall depths in southern Europe are 2.5 times those observed in northern Europe (Poesen and Hooke, 1997). Snowmelt in northern regions or in mountainous areas generates runoff either intermittently through the winter or in the spring.

Topographic, geomorphic and soil characteristics strongly influence the types and location of soil erosion processes in Europe. In the north, on young landscapes primarily composed of glacial and periglacial sediments, particularly cover sands, wind erosion is largely related to strong winds and the presence of dry sandy soils lacking the protection of vegetation. Snowmelt in combination with rain may lead to significant soil erosion by water. Landsliding is a problem on uplifted marine quick clays (e.g. Norway). The loess belt of western and central Europe is a major focus for erosion by water on cultivated land as loess-derived soils with a soil organic matter content less than 2 % rank amongst the most susceptible soils for water erosion in the world (Poesen, 1993). More recently, it has been demonstrated that tillage erosion and erosion due to root and tuber harvesting are also important in this part of Europe. In southern Europe, young, tectonically active areas with strong uplift have resulted in landscapes with a high potential energy. If silt clay deposits (marls) occur, steep slopes are affected by intense mass wasting, water erosion and badland development (e.g. Poesen and Hooke, 1997; Grove and Rackham, 2001; Chapter 2.5). Throughout Europe, coastal areas with sandy and silty deposits (e.g. Bakker *et al*., 1990) may suffer from intense wind erosion. Hence strong geological and pedological controls allied to intensive land use over long periods of time mean that the most erosion-sensitive areas of Europe are the loess belt (with collapsible soils), the marl areas in southern Europe and also the volcanic ash soils in Iceland (Chapter 1.5).

Soil erosion in Europe is, and has been, strongly influenced by land-use change and land policy. In northern and western Europe, post-World War II intensification in agriculture has featured, through land consolidation programmes, remodelling of landscapes in terms of parcel sizes and slope length. It has also led to the abandonment or reduction of mixed farming (mix of cattle and arable farming) with specialisation in livestock farming in some areas and arable farming in others. In the latter case, there has been significant extension of monocultures (e.g. maize). Many of these changes were driven by the EU Common Agricultural Policy (CAP), with the main aim of increasing Europe's self-sufficiency in terms of food production, but this led to over-production and environmental degradation (i.e. soil compaction, surface sealing, soil erosion, muddy flooding and pollution; Bond, 1996). Most governments and agencies are now responding to the increased soil erosion risk (Chapter 2.23). An attempt is being made to reverse this trend through agri-environmental measures (Boardman *et al*., 2003; Chapter 2.24).

In eastern Europe, collectivisation since World War II led to remodelling of landscapes with mechanisation and the creation of large fields; this had a major impact on erosion rates and pollution (e.g. Chapter 1.11). Since 1990, the introduction of free market reforms, return to private ownership and economic decline in agriculture (lack of investment, decline in fertiliser application, etc.) have introduced changes which pose new challenges. Accession to the EU now poses a fresh series of challenges with regard to farming in an environmentally sensitive manner (including limiting erosion) as farming incomes rise and intensification takes place. There is a clear danger of repeating the mistakes of western Europe.

In many hilly areas of Mediterranean Europe, there has been a shift from traditional multiple cropping systems (i.e. contour cultivation of mixed herbaceous and tree crops combined with stabilising underground drainage, contour ditches and terracing on steep slopes, e.g. Coltura Promiscua in central Italy) on small parcels towards monoculture in combination with mechanistaion and up- and downslope cultivation (e.g. vines, almonds) on large parcels (e.g. Chisci, 1986). The loss of traditional landscapes with terraces (e.g. Ambroise *et al*., 1989; Grove and Rackham, 2001) has come about mainly because of a decrease in rural population, a decrease of persons working in agriculture and mechanisation and scale enlargement in

agriculture. Abandonment and disrepair of terraces have in a number of cases led to gully erosion and landsliding. In the Mediterranean, there has also been significant remodelling of badland areas through land levelling (Poesen and Hooke, 1997; Chapter, 2.12) and the creation of terraced landscapes for the establishment of vineyards or greenhouses which do not require soil, only water ('permaculture', 'hydroculture'). EU subsidy systems have encouraged monocultures of olives and almonds at the expense of traditional landscapes (e.g. loss of cork oak forests and expansion of eucalyptus forest). Strong economic incentives to grow certain crops (e.g. grapes) have led to the establishment of cropland on steep, less suitable slopes with a high soil erosion risk (Boardman *et al*., 2003; Chapter 2.12). These changes have had implications for soil erosion and loss of native habitat (e.g. of the lynx in Spain and Portugal).

2.2.4 IS SOIL EROSION A NEW PROBLEM IN EUROPE?

In historical times, soil erosion problems were mainly concentrated in Mediterranean Europe, particularly during the Greek and Roman periods (Vita-Finzi, 1969; van Andel and Zangger, 1990; Bintliff, 1992). The start of the erosion problem strongly relates to woodland clearance and subsequent farming activities. Lang and Bork (2006) cite examples of significant soil erosion due to human impact starting in the Neolithic in the Peloponnesus peninsula and in the Late Bronze Age in the Drama basin of Macedonia (Fuchs *et al*., 2004; Lespez, 2003). Other examples of early human-induced erosion phases in the Mediterranean are reported by Wainwright and Thornes (2004) and Poesen *et al*. (Chapter 2.5). In central and western Europe, the onset of erosion came later and related to the beginnings of arable agriculture. Intense periods of soil erosion have been documented in Germany in medieval times which relate to extreme climatic events. However, it is clear that certain crop types, field and crop patterns and farming practices create landscapes that are sensitive to climatic events, including those of an extreme character (Bork, 1989; Chapter 2.1).

Substantial loss of soil has occurred in the past and modern humans are in many areas cultivating the remnants of a former thick soil cover. This is especially true in the Mediterranean, where much archaeological evidence shows excessive soil loss and impact on Classical civilisations. In northern Europe too, formerly thick loess covers have been substantially lost owing to Historic and Prehistoric farming practices (Favis-Mortlock *et al*., 1997). Hence there is a long legacy of deterioration in soil quality and quantity dating from before to the modern period of intensive farming.

2.2.5 OVERVIEW OF MAJOR SOIL EROSION PROCESSES, THEIR SPATIAL EXTENT, MAJOR CONTROLLING FACTORS AND CONSEQUENCES

2.2.5.1 Soil Erosion by Water

Soil erosion by water (water erosion) comprises sheet or interrill, rill, gully and pipe erosion. Interrill erosion is often preceded by physical soil degradation processes such as soil compaction, surface sealing and crusting (Chapter 2.3 and 2.4). To progress from a noncrusted soil surface state to gullying may take months and requires cumulative rainfall of >450 mm (Papy and Boiffin, 1989) or may occur as a result of a single storm event (Boardman, 1988).

Severe soil erosion by water occurs typically on bare, temporarily unprotected arable land, overgrazed rangelands and on badlands (e.g. De Ploey, 1989).

Cerdan *et al*. (Chapter 2.4) review sheet and rill erosion rates across Europe as measured on experimental runoff plots under different land uses. This allows for a reasonably standardised comparison of rates under different land use conditions. The largest mean sheet and rill erosion rates in Europe have been recorded on

bare soil (23 t ha^{-1} yr^{-1}), vineyards (20 t ha^{-1} yr^{-1}) and maize (14 t ha^{-1} yr^{-1}) whereas shrubland, grassland, orchards and forest typically have values well below 1 t ha^{-1} yr^{-1}. Clearly, erosion rates on runoff plots are not the same as on field parcels with variable lengths and topographies and therefore plot rates may underestimate field rates by large amounts (Evans, 1995).

Poesen *et al*. (Chapter 2.5) conclude that for various reasons, gully erosion has been much less studied in Europe. Compared with sheet and rill erosion rates, however, the limited available data indicate that gully erosion rates are far from negligible and may even exceed 200 t ha^{-1} yr^{-1} in active badland areas of the Mediterranean. The contribution of ephemeral gullies in cropland or permanent gullies in rangeland to total soil loss by surface water erosion may range between 10 and 80 %, depending on the environmental conditions. Compared with sheet and rill erosion, off-site effects of gully erosion may be more important since the development of gully channels in the uplands dramatically increases the connectivity for sediment in the landscape resulting in significant reservoir siltation (Chapter 2.20) and muddy floods (Chapter 2.19).

Faulkner (Chapter 2.6) emphasises the importance of subsurface (pipe and tunnel) erosion processes by water in three distinct European environments: on Histosols and Gleysols in upland, humid northern Europe; in the loess belt; and in the semi-arid Mediterranean basin. In northern Europe, piping is encouraged by discontinuities in the soil profile especially peat overlying mineral soil. In loess, pipes seem to develop along failure planes which focus throughflow into gully heads. Under semi-arid conditions, subsurface flow is related to dispersive, clay-rich soils and sediments which initiate pipe formation and collapse, resulting in gullying and badlands. Despite the importance of pipe erosion in these environments, few or no published data on soil loss rates are available. In some cases, soil losses by pipe erosion may equal or even exceed soil losses by water erosion (e.g. badlands in central Italy; Torri D, personal communication).

In many parts of Europe, rates of soil erosion by water have been on the increase since the middle of the 20th century. The reasons for this vary throughout Europe, but several factors are relevant:

- Efficient weed control in cropland, hence less soil cover and thus more water erosion. Weed control in winter also led to the introduction of winter cereals in the UK in the 1970s, increasing the risk of soil erosion.
- Increase in crop monocultures (e.g. maize, vineyards), leaving the soil unprotected during part of the year.
- Implementation of land consolidation programmes, leading to larger and longer parcels.
- Extensive removal of hedgerows and other types of field boundaries because they are impractical to maintain, expensive and time and labour consuming. This removal has led to a decrease in landscape roughness and buffer capacity to store runoff and sediment. At the same time, the sediment connectivity within these landscapes has increased.
- Movement of arable farming on to steeper slopes as a result of greater vehicle power in the post- World War II period, for instance, ploughing-up of grassland in the UK and ploughing of steep slopes for almond production in Spain (introduction of caterpillar tractors; Poesen *et al*., 1997).
- Intensification of cropland production through the use of chemical fertilisers has led to a decline in organic matter content in soils and loss of their structural stability. The introduction of power harrows has reduced aggregate size of seedbeds (e.g. Speirs and Frost, 1985).

The consequence of these changes has been a significant increase in off-farm impacts, such as eutrophication, phosphate pollution, sediment pollution, muddy floods and reservoir sedimentation. Overall, in the short term the costs related to off-farm impacts seem to be more important than those related to on-farm impacts in Europe.

2.2.5.2 Soil Erosion by Wind

In northern Europe, wind erosion is severe on light, sandy soils (Pleistocene glacial outwash) and on volcanic ash soils (Iceland). In the drier parts of southern Europe, wind erosion also occurs on more silt- or clay-rich

soils, but the problem here is less well researched, and probably less extensive or intense (Warren, 2003). Inappropriate farming practices have increased the wind erosion problem, i.e. enlargement of parcels and removal of hedges, drainage of soils and overgrazing (Chapter 2.7). In contrast to water erosion, very few data on rates of soil erosion by wind in Europe are available.

2.2.5.3 Soil Erosion by Tillage

Although recognised by farmers for many decades (e.g. Weinblum and Stekelmacher, 1963), it is only during the last decade that scientists have studied the intensity and controlling factors of tillage erosion in arable lands of Europe. Reported mean soil erosion rates induced by present-day soil tillage techniques on sloping land in Europe range between 3 and 93 t ha^{-1} yr^{-1} (Chapter 2.9) and are of the same order of magnitude as rates of soil erosion by water. Overall, tillage erosion rates in Europe have increased over recent decades because of an increase in tillage depth and speed (which increases rates of tillage translocation of the plough layer), but also because of the expansion of arable land for crops requiring frequent tillage of the topsoil (e.g. almonds; Poesen *et al*., 1997). Because rills and (ephemeral) gullies in cropland are filled in by soil tillage annually, tillage erosion reinforces soil erosion by concentrated runoff (Poesen *et al*., 2003).

2.2.5.4 Soil Erosion by Land Levelling

Throughout Europe, land levelling has been applied in various regions and this has resulted in significant soil profile truncation: 1 m soil surface lowering represents 15 000 t ha^{-1}. In some cases, the soil surface has been lowered by several metres within less than a year! Hence soil erosion by land levelling can be considered to be the most intense soil erosion process. In addition, land levelling often induces other soil erosion processes such as sheet, rill, gully and pipe erosion, in addition to shallow landsliding resulting in very high soil losses and in significant off-site effects (Chapter 2.12). Despite its importance, soil erosion by land levelling has received limited attention in Europe.

2.2.5.5 Soil Erosion Caused by Crop Harvesting (SLCH)

Over the last two decades, it has become clear that during harvesting of crops such as potato, sugar and fodder beet, chicory and leek, significant amounts of soil (clods, rock fragments and soil adhering to the crop) can be removed from the parcel where these crops are grown. This erosion process, termed SLCH (soil loss due to crop harvesting), is significant in various parts of Europe. Mean SLCH data for Europe range between 2 t ha^{-1} yr^{-1} for potato and 17 t ha^{-1} yr^{-1} for sugar beet (Chapter 2.10). Soil moisture content at harvest time largely controls the magnitude of SLCH in Europe (Ruysschaert *et al*., 2004).

Given its important off-site effects, farmers and the crop processing industry make efforts to reduce SLCH. Nevertheless, this erosion process remains significant and can even be the dominant soil erosion process in flat cropland areas.

2.2.5.6 Shallow Landsliding

Landsliding in general and shallow landsliding in particular occur most frequently on steep slopes (often under rangeland and cropland) with a clay-rich substratum at shallow depth. Shallow landsliding is a soil degradation process in hilly and mountaneous areas of Europe (De Ploey, 1989). Maquaire and Malet (Chapter 2.8) discuss their triggering mechanisms. Although its on- and off-site effects are very significant, limited data on soil losses caused by this process are available.

Most soil erosion problems occur on cultivated land, but also uncultivated land (rangeland, forest) can suffer from significant soil erosion (Evans, 2006). Soil erosion in uncultivated land is often driven by grazing animals (in some cases caused by subsidies increasing sheep numbers and hence stocking rates), by afforestation and drainage, by fire and by the exposure of bare soil through human activities (e.g. increase in soil erosion rates in recreational areas, caused by deforestation, and the establishment of ski resorts, as a result of an increase in disposable income).

On the other hand, in some mountainous areas of Europe (e.g. France, Spain) there has been an increase in forest cover because of depopulation, resulting in a reduction in soil erosion rates.

In conclusion, it should be stressed that several soil erosion processes often operate at the same site. Common soil erosion process combinations in Europe are (1) water erosion (interrill and rill, gully and pipe erosion), tillage erosion and SLCH; (2) wind erosion, water erosion (interrill and rill, gully erosion) and SLCH; and (3) soil erosion by land levelling, water erosion (interrill and rill, gully and pipe erosion), tillage erosion and shallow landsliding. Pan-European soil erosion assessments (Chapter 2.13), available soil erosion datasets (Baade and Rekolainen 2006), soil erosion models (Chapter 2.16) and assessments of the impact of environmental changes on soil erosion across Europe (Chapter 2.18) usually focus on only one or two soil erosion processes, neglecting the other processes. Hence assessments of soil erosion rates for a given area in Europe are often underestimates (Poesen *et al*., 2001). This should be rectified by future soil erosion assessments.

2.2.6 CONCLUSIONS

We have outlined the current understanding of erosion processes as they affect Europe. Gaps in our knowledge remain, for example on the spatial and temporal distribution of various soil erosion processes and their interactions, and these affect our ability to model and predict. It is also clear from the earlier chapters that summarise knowledge in each country that there are great contrasts in the amount of erosion data available. In some countries there are reliable estimates of erosion rates, in others none. This volume is merely a first attempt to draw together existing knowledge and research on the European continent.

Despite the inter-European contrasts, and the continuing need to fill in the gaps, we would argue that action to control soil erosion should continue. There is sufficient knowledge in Europe to apply control techniques and to experiment with the efficacy of those available (including those based on traditional knowledge). Much of the failure to address on- and off-farm impacts of soil erosion is a result not of technical inadequacy, but of a failure to recognise the importance of socio-economic factors in influencing erosion. Erosion often occurs because farmers are encouraged by financial incentives to grow inappropriate crops (or keep animals) on vulnerable sites. The relationship between financial incentives and wise or unwise use of the land is brought to prominence by the recently introduced agri-environmental measures within the EU. The main reason why soil erosion is now a political issue in Europe is that it is beginning to be recognised that it is not simply a farming problem but one with implications for wider civil society. Impacts and costs of erosion are both short and long term, affecting, for example, drinking water quality, freshwater ecosystems and the life of dams.

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2.3

Soil Surface Crusting and Structure Slumping in Europe

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2.3.1 INTRODUCTION

The degradation of soil surface structure by rainfall (surface crusting and structure slumping) has long been recognised to play a significant role in soil erosion. Even though runoff can be generated by low infiltration rate subsurface layers, including ploughpans or frozen subsoil (e.g. Oygarden, 2003), the degradation of soil surface structure often controls runoff triggering. This is especially true under temperate climate where gentle rainfall events (5–10 mm hr^{-1}) could not induce runoff in many soils if the soil surface were not sealed by surface crusts (the infiltration capacity of surface crusts commonly ranges from 0 to 5 mm h^{-1}; Table 2.3.1). In the same way, the role of surface crusting in runoff generation is particularly important in spring and summer when the soil is dry (e.g. Ehlers *et al*., 1980; Dijk and Kwaad, 1996), that is, when runoff is not likely to be generated by soil water logging.

Therefore, erosion risk assessment requires knowledge of the soil, climatic and management conditions that control the various processes involved in soil surface structure degradation. Suggesting relevant management practices also requires (i) diagnostic tools for determining the degradation processes involved in a particular situation and (ii) predictive tests to assess risks of soil surface structure degradation.

Soil surface structure degradation and its impact on erosion have long been studied. In addition to many textbooks, a great deal of information can be found in the proceedings of the three international working

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TABLE 2.3.1 Crust types, subtypes, diagnostic features (according to Valentin and Bresson, 1998) and infiltrability (from Valentin and Bresson, 1992; completed with data from Kwaad and Mullingen, 1991; Fiès and Panini, 1995; Bresson *et al.*, 2001)

Type	Subtype	Main process	Diagnostic features	Infiltrability ($mm\ h^{-1}$)
Structural	Slaking	Aggregate disruption	Thin, dense layer, with sharp lower boundary	1–20
	Infilling	Aggregate erosion and illuviation of eroded particles	Thin, dense layer, with textural separation and rather sharp lower boundary	5–10
	Coalescing	Aggregate deformation	Thick, continuous layer, with convexo-concave voids and progressive lower boundary	2–9
	Agglomerating	Fragment agglomeration	Closely packed agglomerates	No data
	Packing	Particle compaction	Closely packed textural units	25–45
	Sieving	Particle sorting and filtration	Loose sand grains upper layer overlying a thin plasmic layer	0–15
Erosion		Erosion of sieving crusts	Thin plasmic layer at the surface	0–2
Depositional	Runoff	Sedimentation in running water	Poorly sorted micro-bedding	1–5
	Still	Sedimentation in still water	Highly sorted micro-bedding	0–2

meetings which have been held on surface crusting and structure slumping processes, consequences and management: Ghent (Callebaut *et al.*, 1986), Athens (Sumner and Stewart, 1992) and Brisbane (So *et al.*, 1995).

This chapter will deal with (i) a short overview of crusting and slumping processes, contolling factors and consequences, using mainly recent studies carried out in Europe, (ii) surface crusting occurrence in Europe, (iii) structure slumping occurrence in Europe and (iv) a few research opportunities.

2.3.2 CRUSTING AND SLUMPING PROCESSES

2.3.2.1 Definitions

Surface crusting. Two terms can be found in the literature: surface seal and surface crust. Most often, 'seal' refers to water infiltration issues and 'crust' refers to soil strength issues. Therefore, a seal which dries after rainfall becomes a crust, and a crust which rewets under the following rainfall event becomes a seal. Moreover, 'surface sealing' also refers to the consequences of growth in urbanisation and transport infrastructure. Therefore, the term soil 'crust' should rather be used, whether the soil is dry or wet.

A crust is a thin, often transient, soil-surface layer which develops under rainfall or irrigation. A crust usually results from processes induced by wetting and raindrop impact, such as aggregate disruption and/or particle (fragment) relocation and/or compaction. The decreased interaggregate and/or interparticle packing porosity leads to reduced saturated hydraulic conductivity and results in increased strength when dry.

Structure slumping. The term 'slumping' has been suggested for hardsetting soils (Mullins *et al.*, 1990) to distinguish the collapse of seedbeds on wetting from the shrinkage induced by subsequent drying [hardsetting is a soil structure degradation process in which, during drying, the surface horizon sets to a hard, structureless mass that is difficult to cultivate, impedes seedling emergence and restricts root growth (Mullins *et al.*, 1990)]. Slumping

results from processes similar to those involved in crusting (Mullins *et al*., 1990), except that overburden pressure is expected to dominate slumping rather than rainfall kinetic energy (Bresson and Moran, 1995). Although slumping in hardsetting soils has usually been considered through its consequences on crop yields and tillage management, it is also expected to reduce water infiltration rate (Mullins, 1998), all the more because the decrease in macroporosity affects the whole seedbed or tilled layer and not only the top few millimetres.

2.3.2.2 Processes

Surface crusting has been extensively studied in Europe. A conceptual morphological model for soil crusting and slumping has been suggested by Le Bissonnais (1996) and Bresson and Moran (2004). This model includes three main processes: (i) aggregate disruption (abrasion, compression of entrapped air, differential swelling, physico-chemical dispersion) or aggregate deformation, (ii) particle/fragment relocation (infilling, splash, micro-mudflow, micro-deposition) and (iii) compaction (raindrop kinetic energy, capillary forces, overburden pressure).

Crusting has been shown to be a dynamic process, which includes two main development stages (Boiffin, 1986; Valentin, 1986): (i) sealing of the surface by a structural crust and then (ii) development of a depositional crust. The change from the first to the second stage mainly depends on a decrease in infiltration rate due to the structural crust formation, which induces micro-runoff.

The development rate of the structural crust, and also the hydraulic and mechanical properties of the crust, are closely related to the size of the fragments resulting from the aggregate disruption processes (e.g. Le Bissonnais, 1990; Roth and Eggert, 1994). Therefore, structural crust subtypes depend not only on the soil material properties (texture, organic matter content and aggregate stability) but also on the initial water content of the seedbed and on the rainfall characteristics (e.g. Bresson and Cadot, 1992; Le Bissonnais and Bruand, 1993; Fiès and Panini, 1995).

The crust types and the related diagnostic features suggested by Valentin and Bresson (1998) are summarized in Table 2.3.1. Microphytic crusts are not included in this typology, despite their practical significance in soil erosion (e.g. Solé-Bénet et al., 1997; Maestre *et al*., 2002). Indeed, algaes and lichen can colonise any type of crust, so that they should be considered as a particular vegetation cover rather than a particular type of crust (Bresson and Valentin, 1994).

Relationships between crust types and hydraulic properties have been established (e.g. Valentin and Bresson, 1992; Fiès and Panini, 1995), which shows that crust typology may be useful for implementing an expert-based prediction model of soil surface hydraulic behaviour (Table 2.3.1).

Structural slumping results from aggregate dispersion, disruption or deformation (Mullins *et al*., 1987). Slumping processes have been mainly studied in hardsetting soils (e.g. Mullins, 1998) because, in such soils, the structural collapse resulting from wetting greatly controls the hardening on drying (Bresson and Moran, 1995).

Hardsetting soils are common in the tropics but have seldom been described in Europe (Young, 1992). However, a particular subtype of structural crust (Table 2.3.1), called ‘coalescing’ (Bresson and Boiffin, 1990) or ‘aggregate welding’ (Kwaad and Mücher, 1994), and recently described in France and The Netherlands, respectively, was ascribed to aggregate deformation under viscous conditions, that is, a process similar to slumping. In the same way, recent attempts to model the bulk density profiles within structurally degraded seedbeds have associated a crusting component and a slumping component (Bresson *et al*., 2004). Also, agglomeration by wetting of the fine fragments that commonly result from tillage operations in dry soils may result not only in surface crusts but also in slumped surface horizons (Bresson and Moran, 1995). Similarities between crusting and slumping prompts the consideration of microstructure characterization when studying slumping. For instance, Bresson and Moran (2003, 2004) investigated the role played by compaction versus aggregate disruption on seedbed slumping. They showed that aggregate disruption on wetting did not induce

much increase in bulk density but a strong decrease in macroporosity, whereas compaction by either rainfall kinetic energy or overburden pressure led to a strong increase in bulk density.

2.3.2.3 Controlling Factors

The soil, climate and management conditions are well known to control surface structure degradation hazard and rate. Soil characteristics, such as texture (e.g. Fiès and Panini, 1995), clay mineralogy (e.g. Mermut *et al.*, 1995), organic matter content (e.g. Le Bissonnais, 1996) and exchangeable sodium percentage (e.g. Robinson and Philips, 2001), and also slope steepness (e.g. Poesen, 1984) and stone cover (e.g. Poesen and Ingelmo-Sanchez, 1992), play a major role in aggregate stability and therefore in soil susceptibility to surface structure degradation. Other controlling factors depend greatly on management practices: initial conditions such as aggregate size distribution (e.g. Bresson and Moran, 1995) and water content (e.g. Le Bissonnais, 1990), and also soil surface conditions such as surface roughness (Roth and Helming, 1992) and vegetation cover (e.g. Martin, 1999). Climate also plays a great role through rainfall characteristics such as rainfall intensity and kinetic energy (e.g. Helming *et al.*, 1993).

The soil, climate and management conditions also control the type of crust that may form. For instance, on loamy temperate soils, a slaking crust will quickly form if the soil was dry before rainfall, and an infilling crust will slowly develop if the soil was wet (e.g. Bresson and Cadot, 1992). Conversely, on highly unstable silty soils, a coalescing crust usually develops whatever the initial state (e.g. Bresson *et al.*, 2001).

2.3.2.4 Consequences on Erosion

Soil surface structure degradation plays a significant role in Hortonian flow generation, because it leads to a lower infiltration rate, which increases runoff hazards, and to lower surface roughness, which decreases surface detention. Decreased surface roughness also increases flow velocity and therefore the capacity to detach and transport soil particles (e.g. Roth and Helming, 1992).

However, its effects on several erosion subprocesses are ambivalent (Poesen and Govers, 1986). Soil surface structure degradation usually leads to increased soil cohesion, which may eventually lead to lower particle detachment and sediment concentration (e.g. Kwaad and Mullingen, 1991). In cultivated catchments of the northern Paris basin, crusted fields are the main contributors of overall runoff, whereas most of the soil loss comes from freshly tilled, well-structured fields (e.g. Martin, 1999).

2.3.3 SURFACE CRUSTING IN EUROPE

Changing agriculture in the last 50 years has greatly enhanced the occurrence of erosion in cropping systems of western Europe (Monnier and Boiffin, 1986). Intensive agricultural practices and specialisation of large areas in cash crop production has led to lower soil organic matter content. In addition, the increase in acreage of spring crops which do not cover the soil surface in winter has increased the erosion hazards (Martin, 1999). This is especially true in the temperate areas of Europe, where the rainfall intensity is rather low, which means that runoff generation most often requires the infiltration rate to be reduced by surface structure degradation such as crusting and slumping (e.g. Kwaad and Mullingen, 1991).

2.3.3.1 Temperate Areas

Soil surface crusting is common in western Europe, especially on the cultivated silty soils that develop on the widespread loess deposits (Catt, 2001) and that are usually Luvisols, i.e. clay depleted in the upper horizons.

This might explain why most studies on crusting were carried out in the European loess belt: Belgium (e.g. De Ploey and Mücher, 1981; Poesen and Govers, 1986), Croatia (Racz, 1986; Kisic *et al*., 2002), Finland (Yli-Halla *et al*., 1986), France (e.g. Bresson and Boiffin, 1990; Le Bissonnais, 1990; Auzet *et al*., 1995), Germany (e.g. Ehlers *et al*., 1980; Gross and Tebrugge, 1992; Roth and Eggert, 1994), Hungary (Varallyay and Lesztak, 1990), The Netherlands (e.g. Imeson and Kwaad, 1990; Kwaad and Mücher, 1994), Sweden (Stenberg *et al*., 1995) and UK (e.g. Boardman and Hazelden, 1986). Nevertheless, crusting has also been described on other light-textured soils, namely soils developed on glacial and periglacial deposits (Yli-Halla *et al*., 1986; Vensteelant *et al*., 1997; Roth, 1995), alluvium (Gross and Tebrugge, 1992) and sandy drifts (Gross and Tebrugge, 1992). Sodic soils, which are extremely prone to soil structure degradation through physico-chemical dispersion, are much more common in central Europe than in western Europe. However, few studies dealing with surface crusting in sodic environments have been published in international journals, except in Hungary (Varallyay and Lesztak, 1990).

In many studies which deal with erosion processes, soil surface crusting is often simply cited as a cause of runoff generation, but not described and even less discussed. Usually, depositional crusts can be identified in the literature, but the various types of structural crusts can seldom be determined on the basis of the data provided. Slaking crusts seem to be common in most soils, whereas coalescing crusts (Figure 2.3.1d) are described only in the most light-textured, unstable soils (Kwaad and Mücher, 1994; Bresson and Boiffin, 1990; Bresson *et al*., 2001).

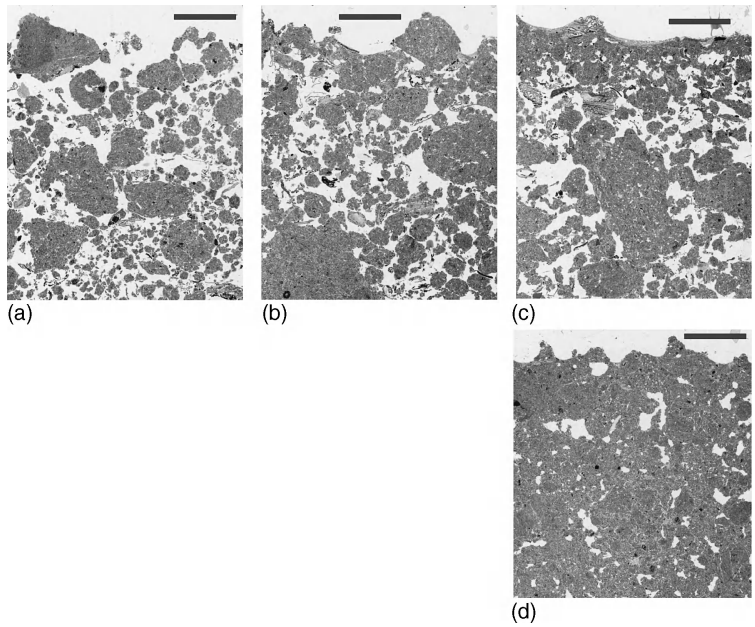

Figure 2.3.1 Surface structure degradation of a Typic Hapludalf developed on a silty loam loess deposit in the Paris basin (reconstructed seedbeds exposed to a 19 mm h^{-1} simulated rainfall). Soil amended with urban waste compost: (a) initial structure; (b) incipient structural crust (infilling) after 5 mm of rainfall; (c) incipient depositional crust developping on a structural crust after 19 mm of rainfall. Untreated soil: (d) slumped seedbed after 19 mm of rainfall. (Vertical thin sections, plain light, scale bar 10 mm)

2.3.3.2 Mediterranean Areas

In the Mediterranean areas of Europe, rainfall intensity may be high enough to trigger surface runoff whether the soil surface is degraded or not (e.g. Uson and Ramos, 2001). However, because of the higher rainfall kinetic energy, scarcer vegetation cover and lower soil organic matter content, surface structure degradation commonly occurs on most soil materials, which greatly enhances surface runoff and subsequent erosion (e.g. Ramos *et al*., 2000). For the same reasons, surface crusting also occurs in noncultivated situations (e.g. forests, rangelands and steppes). Silty and loamy soil materials, which are prone to aggregate slaking, are especially affected by crusting (e.g. Léonard and Andrieux, 1998; Ramos *et al*., 2000), but crusts also develop on a wide range of more stable materials such as clay materials: black marls (Malet *et al*., 2003) and molasse (Boudjemline *et al*., 1993; Léonard and Andrieux, 1998) in southern France, schists in Portugal (Shainberg *et al*., 1991) and Spain (Valcarcel *et al*., 2003) and alluvium in Portugal (Shainberg *et al*., 1991), Italy (Pagliai *et al*., 1995) and France (Léonard and Andrieux, 1998). Several studies in Spain (Solé-Benet *et al*., 1997; Canton *et al*., 2001) and Italy (Robinson and Philips, 2001) have dealt with badlands where crusts were shown to enhance runoff.

As opposed to temperate areas, sodic soils which are sensitive to structure degradation through physico-chemical dispersion are widespread in Mediterranean areas in Europe. However, most studies dealing with sodic soils directly relate surface runoff and erosion to the ESP or the dispersibility of the soil material, and do not attempt to characterise crust development or crust hydraulic properties (e.g. Robinson and Philips, 2001). As pointed out by Bresson and Valentin (1994), crust development in clayey and sodic environments should be related to swelling and cohesion rather than to the physico-chemical dispersability *sensu stricto*.

Only in a few papers can the crust type be identified or inferred from the data provided, with the exception of depositional crusts, which are easily recognized. In cultivated soils, e.g. soils under viticulture, slaking crusts seem to be common, but coalescing crusts have also been described in nonsodic loamy soils (Uson and Poch, 2000).

2.3.3.3 Relationships Between Crust Types and Climate

Microphytic crusts have been extensively studied in arid and semi-arid climates, especially in the USA, Australia and Africa. Such crusts have also been observed in arid and semi-arid areas of Mediterranean Europe (e.g. Solé-Benet *et al*., 1997; Maestre *et al*., 2002) and in temperate areas of Europe (Pluis and de Winder, 1989). This means that the abundance of a particular type of crust under specific environmental conditions does not necessarily imply that such a crust type cannot develop elsewhere.

Whatever the climate, sandy soils may also be affected by crusting. In these soils, a particular type of structural crust develops (‘sieving’ crust, Table 2.3.1), that has been extensively studied in semi-arid intertropical areas (e.g. Valentin, 1986; Bielders and Baveye, 1995). In temperate climates, however, only a few studies have been devoted to crusts developed on sandy soils (Valentin and Bresson, 1998; Larue, 2001). This lack of interest might be due to the low fertility potential of such soils. Sandy materials usually lead to poor, acidic soils which are covered by forest and meadows where crusting is not expected to occur. If cropped, these soils are usually affected by severe slumping and compaction processes, so that crusting might not appear to be the main structure degradation problem.

2.3.3.4 Surface Cruting Sensitivity Map of Europe

From the above review of international scientific journals and databases, it appears that surface crusting has been studied on a rather small number of sites. In order to overcome the lack of geographic information on the occurrence of this process, a map of crusting sensitivity (Figure 2.3.2), based on the Soil Geographical Data Base of Europe, has been suggested (Le Bissonnais *et al*., 2005). In this study, crusting sensitivity is

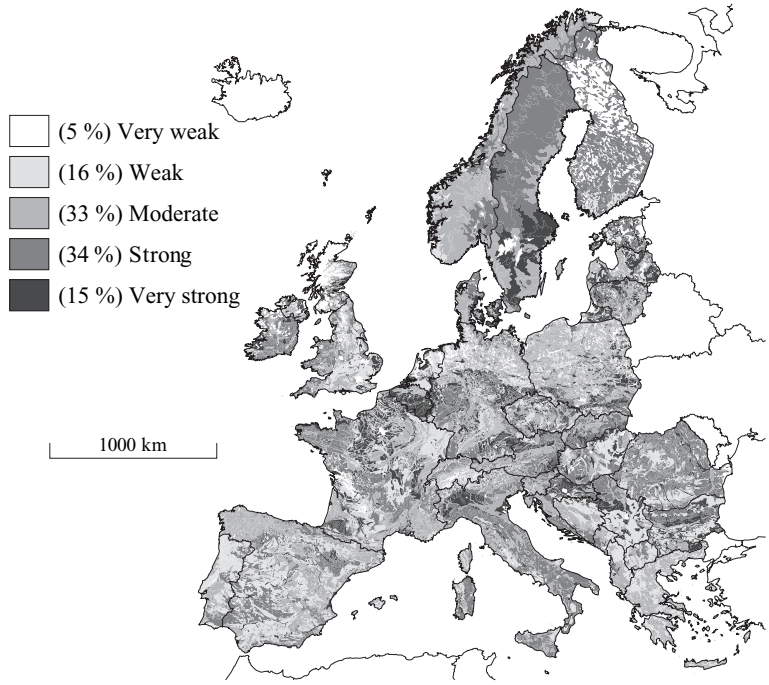

Figure 2.3.2 Soil crusting sensitivity map of Europe

characterised using two parameters. The textural parameter comes from the dominant soil surface texture (described by five classes in the database). The physico-chemical parameter is derived from the soil name at the third classification level, by taking into account the positive or negative effect of organic matter content, exchangeable sodium percentage, carbonates and other pedogenetic characteristics on structural stability. Because only few soil parameters are explicitly present in the Soil Geographical Data Base of Europe, the pedotransfer rules used in this expert-based model of crusting sensitivity are rather rough. However, they are consistent with the current knowledge of the processes involved in soil surface crusting. Therefore, this map may constitute an interesting guide for further investigations on the occurrence of soil surface crusting in Europe.

2.3.4 STRUCTURE SLUMPING IN EUROPE

Although slumping has mainly been studied in hardsetting soils, which are widespread in the tropics, it is expected to occur in unstable, sandy soils of most climatic zones, including the temperate and Mediterranean zones (Mullins *et al*., 1990). In Europe, it has mainly been studied in the UK, on sandy loam soils with low organic matter content (Young *et al*., 1991; Young, 1992), where slumping and compaction might not be easily delineated (Young, 1992). Only a few references can be found to other European countries: in The Netherlands (Kwaad and Mücher, 1994), Sweden (Stenberg *et al*., 1995) and France (Figure 2.3.1d) (Bresson *et al*., 2001).

The typology of soil surface characteristics suggested by Léonard and Andrieux (1998) includes both the surface crust and the underlying tilled layer. This means that slumping and/or compaction of the layers

underlying the crust played a significant role in the surface hydraulic properties of the light-textured soils in the wine-growing Mediterranean catchment that they studied. In their study on the effect of surface roughness and structure on runoff generation, Gascuel-Odoux *et al*. (1991) also provided evidence of a compacted horizon underlying the depositional crusts. Moreover, the global slumping of seedbeds is often recognized by farmers, and such a process is commonly called in French 'prise en masse' (Boiffin and Sébillotte, 1976). Therefore, slumping is likely to be rather common in most light-textured soils of Europe. It is surprising that only a few papers devoted to slumping in Europe have been published in international soil science journals. Given that slumping soils are also prone to crusting under rainfall and to compaction under tillage operations and machinery traffic, the lack of publications on slumping may reflect the fact that crusting and compaction were considered as a more important issue for these soils than slumping.

2.3.5 CONCLUSIONS

In temperate areas of Europe, erosion mainly occurs on cultivated silty and loamy soils developed on loess deposits because of surface crusting, seedbed slumping or compaction of subsurface horizons. In Mediterranean areas, soil surface structure degradation is widespread and significantly increases erosion hazards. This prompted European agronomists and soil scientists to study the processes involved in order to establish relevant diagnostic tools, predictive tests, management practices and models.

In the last 10 years, i.e. since the last international working meeting on soil crusting and slumping, most studies carried out in Europe have dealt with five main issues: (i) validation of a process-based test for aggregate stability that could be used as a predictive tool (e.g. Le Bissonnais, 1996; Fox and Le Bissonnais, 1998), (ii) improvement of a comprehensive typology of crusts which could be used as a diagnostic tool (e.g. Valentin and Bresson, 1998), (iii) incorporation of soil surface characteristics (crust morphology, surface cover, surface roughness, etc.) in runoff and erosion studies (e.g. Auzet *et al*., 1995; van Wesemael *et al.*, 1996; Léonard and Andrieux, 1998), (iv) modelling crust hydraulic conductivity (e.g. Burt, 1998; Vandervaere *et al*., 1998) and (v) incorporation of crusting in soil erosion models (e.g. De Roo *et al*., 1996; Le Bissonnais *et al*., 1998; Cerdan *et al*., 2002).

Some suggestions for further studies arise from this brief overview:

1. *Crusting and slumping occurrence in Europe* To overcome the lack of geographical information on the occurrence of soil crusting and slumping in Europe, studying the national literature (journals, reports) might be helpful. Using indirect assessment techniques such as remote sensing (e.g. Mathieu *et al*., 1997; De Jong *et al*., 1999) should also significantly improve the proposed crusting sensitivity map.

2. *From conceptual crusting models to crust modelling* Combining a crust morpho-genetic typology with a process-based stability test should lead to a relevant process-based model for soil surface crust development. Whatever the model, expert-based or physically based (Le Bissonnais, 1990; Panini *et al*., 1997; Roth, 1997), more quantitative data dealing with the relationships between crust development and soil material properties, initial conditions, climatic conditions and management practices will be required.

3. *Accounting for spatial and temporal variability* Spatial variability of soil surface conditions has been shown to be very important in runoff and erosion. Surface conditions include not only the crust type and abundance but also other features such as surface roughness, soil cover (vegetation, litter, stones), surface macropores (cracks, channels) and wheel tracks (e.g. Poesen and Ingelmo-Sanchez, 1992; Auzet *et al*., 1995; Léonard and Andrieux, 1998; Cerdan *et al*., 2002; Malet *et al*., 2003). However, assessment methods still need to be improved.

Most studies on crust development have been focused on the earlier stages of seedbed structural evolution. Few studies have included the evolution of surface crusts with successive rainfalls and throughout the cropping season (Kwaad and Mullingen, 1991; Roth and Helming, 1992; Diekkrüger and Bork, 1994; Fohrer *et al*., 1999).

Eventually, this will lead to improved incorporation of crusting and slumping processes into erosion models.

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2.4

Sheet and Rill Erosion

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2.4.1 INTRODUCTION

Water erosion is commonly divided into different subprocesses. This chapter focuses on erosion processes ranging from sheet (or interrill) erosion, which consists of the removal of a fairly uniform layer of soil by raindrop splash and sheet flow, to rill erosion, which results in the formation of numerous and randomly occurring small channels of only several centimeters depth under the action of small, intermittent water courses usually also only several centimeters deep (Glossary of Soil Science Terms, http://www.soils.org/sssagloss). To measure the rates and extent of sheet and rill erosion, both indirect and direct methods have been used. Indirect methods generally measure soil profile truncation or sediment accumulation relative to a reference soil horizon, to an exposed or buried reference object (exposed or buried roots, foundations, etc.), or to the loss or accumulation of tracers. These methods are more appropriate for studying historical erosion. To

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assess current sheet and rill erosion rates, direct methods, mainly plot or catchment monitoring and field-based measurements (e.g. mapping of erosion features) are reported. Field-based methods are most effective to answer questions such as, where does linear erosion occur and is it a problem? However, they cannot properly monitor sheet erosion and, more important for this study, their applications have been restricted to very few places in Europe. The best available data to compare soil erosion rates in Europe induced by sheet and rill processes come from plot measurements. These represent relatively well-standardized data, which can give reliable information on slope sensitivity to sheet and rill erosion under a given set of conditions, and they are widespread. Based on a large dataset of soil erosion measurements under natural rainfall at the plot scale, the objectives of this study were (i) to quantify the different sheet and rill erosion rates in various agro-environmental settings throughout western and central Europe, (ii) to identify the more at-risk situations in terms of land use or physiographic conditions and (iii) to assess overall sheet and rill erosion rates for Europe.

2.4.2 METHODOLOGY

An extensive database of short- to medium-term (1–10 years) soil loss measurements at the plot scale was compiled from the literature. This database contains 208 entries (one entry corresponds to the combination of one land use, slope, etc., for one experimental site) distributed among 57 experimental sites in 13 countries, representing a total of 2162 plot-years. Only data from experiments with a direct measurement of soil erosion rates, i.e. with an experimental device to measure erosion during natural rainfall events, were collected (e.g. collecting tanks or tipping buckets with or without automatic samplers). On average, the experiments cover $\sim$10 equivalent (eq.) plot-years with a median of 6 plot-years per entry, the maximum being for cereal plots in Portugal (96 eq. plot-years; Lopes *et al.*, 2002) and in Germany, where bare plots have been monitored for 60 eq. plot-years (Martin, 1988; Auerswald, 1993). As shown in Figure 2.4.1 and Table 2.4.1, the database is composed of sheet and rill erosion rate measurements from Austria, Belgium, Denmark, France, Germany, Greece, Italy, Lithuania, The Netherlands, Portugal, Spain, Switzerland and the UK. The corresponding annual rainfall in the database range from $<$200 mm (Spain) to $>$1300 mm (Germany), with a median annual value of 595 mm. No restriction regarding slope length was made when selecting the experimental sites as long as the land use was uniform. However, the median size of the plots is close to the Wischmeier plots with a median slope length of 20 m, a median area of ca 60 m^2 and a median slope of 13.2 % (94 and 75 % of the entries have a slope length $>$5 and 9 m, respectively, which are two recognised thresholds for rill initiation and development).

To compile the database, data with a similar location, land use, slope, slope length, area and soil texture (five classes) were aggregated (weighted for plot years of measurements). As a consequence, data were combined even if other parameters that influence the erosion response were different, for example, data showing differences in soil types or soil surface properties which are not reflected in the textural classification used, difference in tillage systems or direction (parallel or perpendicular to the contour) or differences in slope aspects. Experimental data where a strong evolution with time (e.g. Francia *et al.*, 2002) was reported were not included in the database, as it was difficult to calculate a relevant mean value.

2.4.3 RESULTS AND DISCUSSION

The mean sheet and rill erosion rates are presented in Table 2.4.2 and Figure 2.4.2. The erosion responses between the different land use classes differ significantly (Kruskal-Wallis test statistics = 79.1 with probability $<$0.0001). If we rank (in descending order) the land use classes with at least 25 eq. plot-years of measurement according to the observed sheet and rill erosion rates, we obtain bare soil, vineyard, maize, spring crops, cereal, post-fire, forage, shrubs, grassland and forest. Bare soil is the most represented class with 563 eq. plot-years and, with the vineyard class, have the highest mean rates (23.4 and 20 t ha^{-1} yr^{-1} respectively). Maize

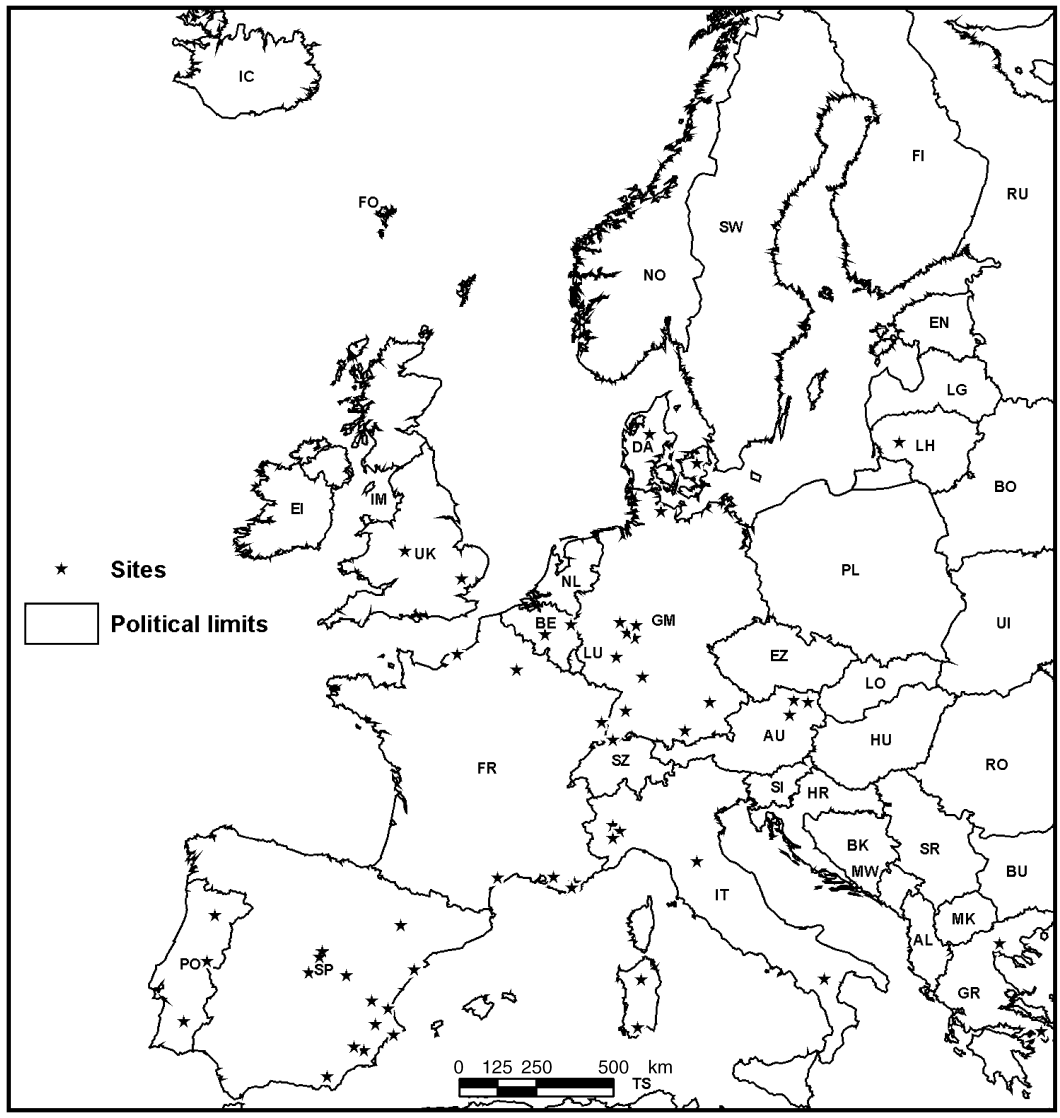

Figure 2.4.1 Location of the experimental sites used in this study

and spring crops also show very high rates, i.e. more than $10\,t\,ha^{-1}\,yr^{-1}$. Interestingly, spring crops have the highest mean yearly rainfall amount (749 mm) and a relatively low mean yearly runoff ($\sim$16 mm), which also imply high sediment concentration. Cereal, post-fire and forage have moderate rates of $\sim 1.5\,t\,ha^{-1}\,yr^{-1}$. Despite relatively steep slopes, the classes shrubs, grassland and forest have the lowest rates, i.e. $<1\,t\,ha^{-1}\,yr^{-1}$ and have relatively high mean yearly runoff volumes, which, conversely to spring crops, imply very low sediment concentration. In fact, land uses with the highest percentage of bare soil, either spatially (wide interrow length and low leaf cover, e.g. vineyard or maize) or temporally (long intercrop duration, e.g. maize or spring crop) have the highest rates. The assemblage of these plot data in a database

TABLE 2.4.1 Description of the soil erosion plot database

Country	No. of entries[a]	Total eq. plot-year	Mean eq. plot-year	References
Austria	8	43	5	Klik *et al.*, 2001; Klik, 2003
Belgium	3	31	10	Bollinne, 1982
Denmark	6	16	3	Veihe and Hasholt, Chapter 1.4
France	11	59	5	Viguier, 1993; Messer, 1980; Martin *et al.*, 1997; Bissonnais 2004; Cerdan *et al.*, 2002; Clauzon and Vaudour, 1971
Germany	41	400	10	Martin, 1988; Auerswald, 1993; Goeck, 1989; Goeck and Geisler, 1989; Dikau, 1986; Voss, 1978; Emde, 1992; Jung and Brechtel, 1980
Greece	8	48	6	Kosmas *et al.*, 1996; Romero-Diaz *et al.*, 1999; Diamantopoulos *et al.*, 1996
Italy	33	433	13	Tropeano, 1983; Zanchi, 1983; 1988; Rivoira *et al.*, 1989; Porqueddu and Roggero, 1994; Caredda *et al.*, 1997; Vacca *et al.*, 2000; Basso *et al*., 2002
Lithuania	11	134	12	Jankauskas and Jankauskiene, 2003
The Netherlands	3	35	12	Kwaad, 1991; 1994; Kwaad *et al.*, 1998
Portugal	16	482	30	Roxo, *et al.* 1996; Figueiredo *et al.*, 1998; Lopes *et al.*, 2002
Spain	48	367	8	Andreu *et al.*, 1994 cited by Cerdà 2001; Bautista *et al.*, 1996; Bautista, 1999 cited by Cerdà, 2001; Andreu *et al.*, 1998a & b; Andreu *et al.*, 2001; Sirvent *et al.*, 1997; La Roca, 1984 cited by Cerdà 2001; Castillo *et al.*, 1997; Puigdefabregas *et al.*, 1996; Padron *et al.*, 1998; Romero-Diaz *et al.*, 1999; Lopez-Bermudez *et al.*, 1991; 1998; Canton *et al.*, 2001; Nicolau *et al.*, 2002
Switzerland	2	9	5	Schmidt, 1979
UK	18	104	6	Fullen and Reed, 1986; Fullen, 1991; 1992; Quinton, 1994

[a]One entry corresponds to the combination of one land use, slope, etc., for one experimental site.

allows comparison of the impact of very different land uses in a common framework and thus confirm ideas that were commonly assumed about the sensitivity of certain crops (e.g. maize, spring crops) to sheet and rill erosion. However, as always with results directly deduced from an experimental dataset, the limits concerning the representativeness of this database should be questioned. Two types of limits can be highlighted:

1. *Limits related to spatial representativeness*

Even if the database is rather extensive, good-quality long-term plot data are not available for every agro-ecological zone in Europe. For example, values up to 200 t ha^{-1} per rainfall event were observed in southwest France (Le Bissonnais *et al.*, 2003) for high-intensity storms on agricultural areas with low vegetation density, but no long-term plot studies have ever been conducted in this area (some possibly high-risk crops as hop or vegetable are also missing from the plot database). Some plot studies are set up systematically according to predefined large monitoring schemes (e.g. Wischmeier plots with different soil types, tillage systems or crop rotations) whatever the erosion risk and are therefore rather objective. On the other hand, other plot studies might stress a particularly at-risk area. In the latter case, extrapolation of results without careful attention to the specificity of the site can lead to an overestimation of the problem.

TABLE 2.4.2 Description of the soil erosion database aggregated according to land use

Land use	No. of entries[a]	Equivalent plot-year	Mean area (m^2)	Mean slope (%)	Mean rainfall ($mm\ yr^{-1}$)	Mean slope length (m)	Mean runoff ($mm\ yr^{-1}$)	Mean erosion ($t\ ha^{-1}\ yr^{-1}$)
Bare soil	54	563	60.0	15.9	674	14.4	91.7	23.40
Vineyard	10	113	100.3	19.3	629	52.3	80.8	19.97
Maize	6	27	38.1	9.9	676	12.9	63.7	13.95
Spring crop[b]	13	62	375.8	11.0	749	43.4	16.1	10.64
Maize + cover	3	21	21.2	8.7	560	10.9	19.9	2.65
Cereal	36	335	1641.2	12.7	629	37.7	19.3	2.10
Post-fire	8	112	1859.0	28.7	466	11.3	40.2	1.54
Forage	9	192	500.2	17.3	661	34.7	27.6	1.35
Vineyard + grass	5	12	102.5	24.0	598	62.7	17.1	0.78
Arable crops	6	139	16.0	10.8	862	8.0	42.2	0.53
Shrubs	34	283	65.3	22.1	411	16.2	9.5	0.50
Grassland	16	231	179.5	15.9	623	31.5	15.2	0.29
Barley + cover	1	3	66.3	10.0	665	22.1	—	0.28
Forest	6	51	48.7	19.9	483	11.8	6.0	0.10
Orchard	1	18	30.0	19.0	467	10.0	0.7	0.05
Total/mean	208	2162	466	16.4	609	25.7	41.0	8.76

[a]One entry is the combination of one land use, slope, etc., for one experimental site.
[b]Except maize, maize + cover and barley + cover classes.

2. *Limits related to the representativeness of the sheet and rill erosion processes*

Erosion is a scale-dependent process hence, depending on the size of the monitoring schemes, results differ, one reason being the influence of slope length, relief patterns or the spatial variability in soil surface conditions on the balance between sediment transport and deposition. Plot studies, being limited in space, will therefore

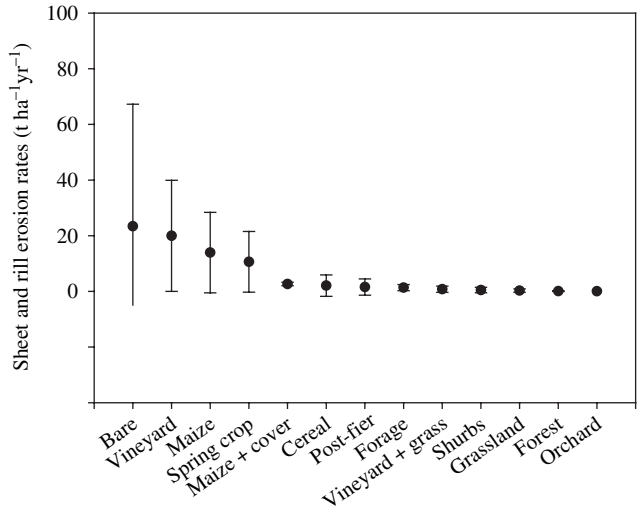

Figure 2.4.2 Mean (±SD) sheet and rill erosion rates aggregated per land use

not reflect everything about what is happening in the landscape in terms of sheet and rill erosion. The results should be understood as a comparison of the sensitivity of given slopes to sheet and rill erosion in a given set of conditions. However, whether the observed soil losses will leave the field or catchment where they originate or will be deposited needs to be addressed through further investigations. On most soil loss plots, the combined effect of interrill and rill erosion is measured. Most of the time, the relative contribution of rill against sheet erosion to the total soil loss remains unknown.

2.4.3.1 Geographical Distribution of Soil Losses

Numerous observations cover the Mediterranean zone, 1382 eq. plot-years data for 113 entries against 780 plot-years for 95 entries for the rest of Europe (Table 2.4.3). Overall, the sheet and rill erosion rates for the Mediterranean zone (MZ) are comparable to those of the rest of Europe.

In the MZ, rates are higher for bare soils ($\sim$32 t ha^{-1} yr^{-1} for the MZ against 17.3 t ha^{-1} yr^{-1} for the rest of Europe) but lower for most of the crop types, although the slopes are steeper. One possible explanation for these differences in the mean sheet and rill erosion rates for the crops (e.g. 0.7 t ha^{-1} yr^{-1} for cereals in the MZ

TABLE 2.4.3 Description of the soil erosion database aggregated according to location and land use

Zone	Land use	No. of entries	Equivalent plot-year	Mean area (m^2)	Mean slope length (m)	Mean slope (%)	Mean rainfall (mm yr^{-1})	Mean runoff (mm yr^{-1})	Mean erosion (t ha^{-1} yr^{-1})
Mediterranean	Bare soil	23	246	113.9	21.0	18.0	559	90.6	31.62
	Vineyard	6	101	99.4	32.2	16.4	640	116.4	16.64
	Vineyard + grass	2	6	100.0	41.8	23.5	582	33.3	1.92
	Post-fire	8	112	1859.0	11.3	28.7	466	40.2	1.54
	Forage	9	192	500.2	34.7	17.3	661	27.6	1.35
	Cereal	18	244	222.6	22.8	13.8	520	24.7	0.66
	Shrubs	31	275	70.1	17.0	22.1	375	9.4	0.54
	Grassland	11	142	180.8	22.9	15.6	564	16.9	0.42
	Forest	4	46	65.0	13.8	19.9	334	8.6	0.15
	Orchard	1	18	30.0	10.0	19.0	467	0.7	0.05
	Total/Mean	113	1382	281.2	21.7	18.7	500	39.8	7.87
Other	Vineyard	4	12	105.0	82.5	23.8	612	21.4	24.96
	Bare soil	31	317	21.7	10.1	14.4	760	93.2	17.30
	Maize	6	27	38.1	12.9	9.9	676	63.7	13.95
	Spring crop[a]	13	62	375.8	43.4	11.0	749	16.1	10.64
	Cereal	18	91	3059.9	52.6	11.6	739	11.6	3.53
	Maize + cover	3	21	21.2	10.9	8.7	560	19.9	2.65
	Arable crops	6	139	16.0	8.0	10.8	862	42.2	0.53
	Barley + cover	1	3	66.3	22.1	10.0	665	—	0.28
	Shrubs	3	8	16.0	8.0	—	780	10.8	0.13
	Vineyard + grass	3	6	105.0	76.7	24.3	608	1.0	0.02
	Grassland	5	89	176.7	50.4	16.3	751	0.7	0.01
	Forest	2	5	16.0	8.0	—	780	0.7	0.003
	Total/Mean	95	780	691.8	30.1	13.4	738	43.0	9.83
Grand total		208	2162	466	25.7	16.4	609	41.0	8.76

[a]Except maize, maize + cover and barley + cover classes.

against 3.5 t ha^{-1} yr^{-1} for cereals in the rest of Europe) is the high rock fragment content found in the MZ soils (e.g. Poesen and Lavee, 1994; Puigdefabregas *et al.*, 1996). The influence of surface stoniness on the decrease of sheet and rill erosion rates is described in many studies (see, for example, the references for Spain in Table 2.4.1) and percentage stone covers of 30–50% are regularly observed. Rates are also higher in the MZ for permanent cover such as grasslands, forests or shrubs, which can probably be related to differences in vegetation density for these land uses in the two zones, natural or perennial vegetation being less dense and with species having lower leaf cover in the MZ.

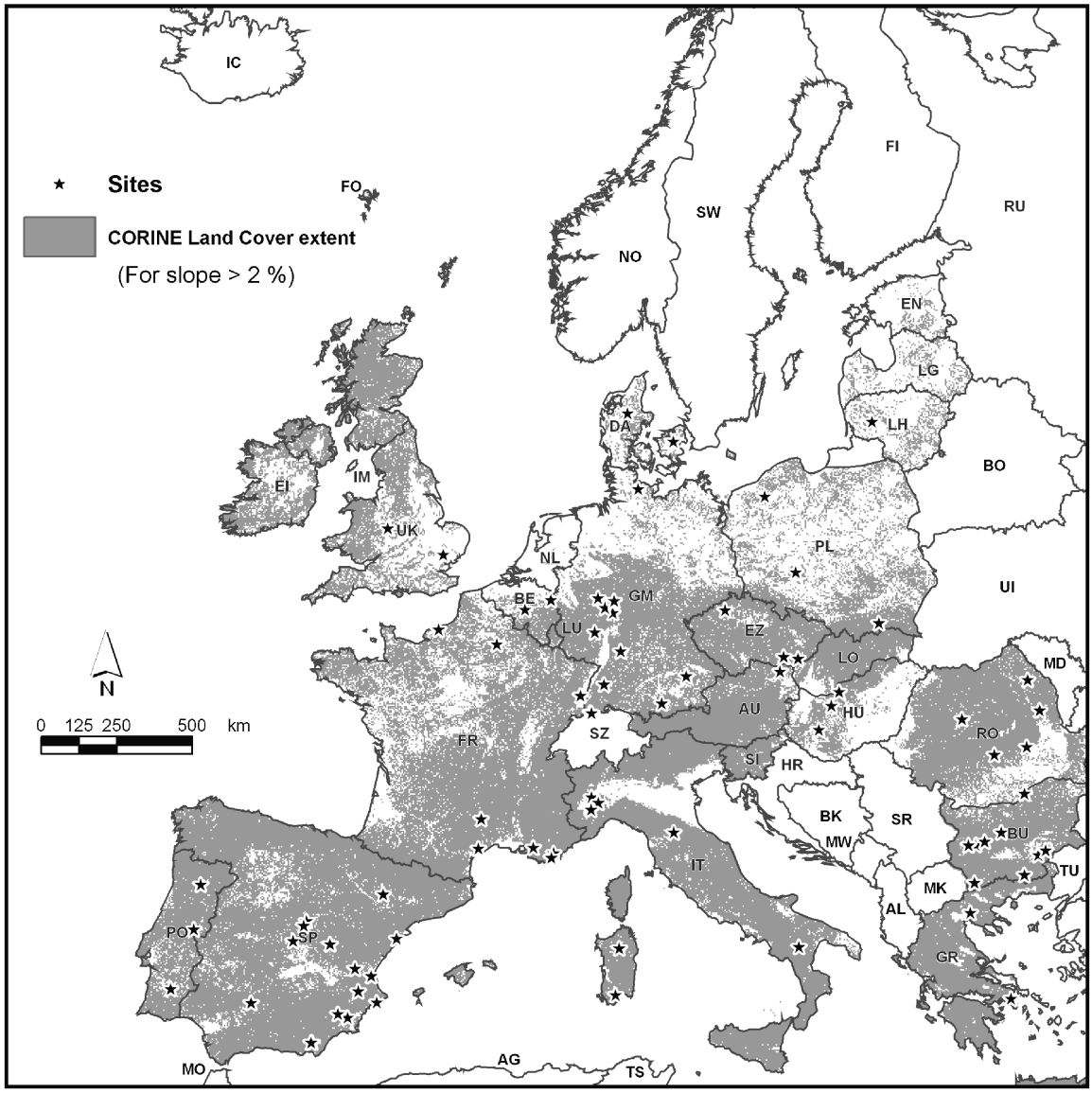

Figure 2.4.3 Extent of the reclassified CORINE land cover classes used in this study (areas with slopes below 2 % or outside the CORINE extent are represented in white)

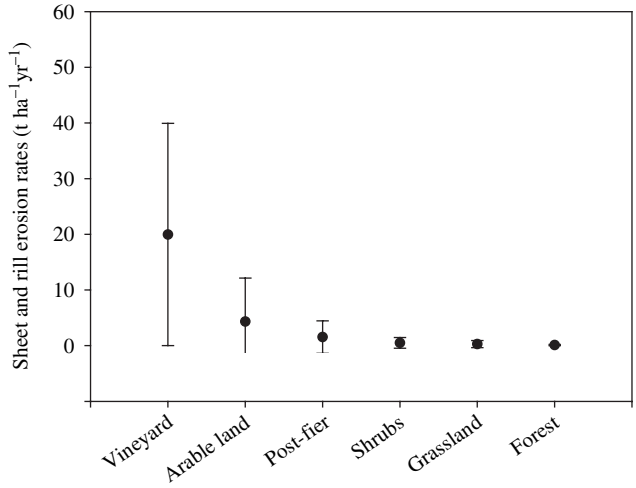

Figure 2.4.4 Mean (± SD) sheet and rill erosion rates aggregated per land uses present in the reclassified CORINE land cover classes

2.4.3.2 Extrapolation of Experimental Data to Europe

Mean sheet and rill erosion rates differ significantly according to land use. It is therefore interesting to calculate the spatial extent of the different land uses to assess sheet and rill erosion rates for Europe. Land cover can be estimated for Europe from the CORINE database. To homogenise the land use classes between the CORINE database and the soil loss database, both databases were reclassified. Figure 2.4.3 presents the extent of the reclassified CORINE land cover classes used in this study (the area where the slopes are <2 % are omitted as corresponding soil losses are usually very small) and Figure 2.4.4 and Table 2.4.4 present the soil loss database aggregated according to the reclassified CORINE land covers. Table 2.4.5 presents the potential mean sheet and rill erosion per land use according to its extent and erosion rate.

It is interesting that arable lands produce ~70 % of total soil loss. The mean calculated sheet and rill erosion rates for Europe are ~1 t ha^{-1} yr^{-1} for the total area and ~1.6 t ha^{-1} yr^{-1} for the erodible areas (i.e. in Table 2.4.4,

TABLE 2.4.4 Description of the soil erosion database aggregated according to the reclassified CORINE land covers

Land use	No. of entries	Equivalent plot-year	Mean area (m^2)	Mean slope length (m)	Mean slope (%)	Mean rainfall (mm yr^{-1})	Mean runoff (mm yr^{-1})	Mean erosion (t ha^{-1} yr^{-1})
Bare soil	54	563	60	14.4	15.9	674	91.7	23.40
Vineyard	10	113	100	52.3	19.3	629	80.8	19.97
Arable land	74	779	931	32.6	12.4	674	25.9	4.34
Post-fire	8	112	1859	11.3	28.7	466	40.2	1.54
Vineyard + grass	5	12	102	62.7	24.0	598	17.1	0.78
Shrubs	34	283	65	16.2	22.1	411	9.5	0.50
Grassland	16	231	179	31.5	15.8	623	15.2	0.29
Forest	6	51	49	11.8	19.9	483	6.0	0.10
Orchard	1	18	30	10.0	19.0	467	0.7	0.05
Total/mean	208	2162	466	25.7	16.4	609	41.0	8.76

TABLE 2.4.5 Mean sheet and rill erosion amounts and rates[a] for the reclassified CORINE land covers

Land use	Area (10^4 ha)	Mean sheet and rill erosion rates (t ha^{-1}yr^{-1})	Mean sheet and rill erosion (10^4 t yr^{-1})	Mean slope (area <2 % excluded)	Mean slope (%)
No soil	1410	0	0.0	21.7	13.9
Arable land	5515	4.337	23919	6.7	3.9
Rice fields	7	0	0.0	4.7	1.1
Vineyards	292	19.97	5832	9.4	7.5
Orchards	518	0.052	27	15.3	13.6
Complex cultivation	3617	0.502	1816	11.4	8.6
Forest	6498	0.1	650	20.6	15.8
Grassland	3212	0.289	928	15.6	10.6
Shrubs	2415	0.502	1212	23.1	21.1
Post-fire	22	1.541	35	20.8	20.3
Wetland	127	0	0.0	12.2	6.4
Slope <2 %	11351	0	0.0	<2	<2
Total	34983	—	34418		

[a]Mean erosion rate: for the total surface ca. 1 t ha^{-1} yr^{-1}, for the erodible areas 1.6 t ha^{-1} yr^{-1}s.

land uses with a sheet and rill erosion rate >0). These mean values are, however, not an indicator of the significance of soil erosion in Europe as they average out spatial variabilities. For arable land in general and more specifically for vineyards (~20 t ha^{-1} yr^{-1}) and spring crops (~12 t ha^{-1} yr^{-1}), the average rates are well above acceptable rates of soil erosion (i.e. rates of erosion exceed rates of soil production). From our calculations, it appears that at least 16.7% of the total area covered by CORINE suffers from significant soil erosion problems. These figures are only indicative and should not be taken as absolute values. Furthermore, in addition of the approximation related to the extrapolation of plot data, the mean sheet and rill erosion rates should be corrected for mean slope (Figure 2.4.5), particularly for arable land (12.4 % for the plot database against an estimated 6.7 % for CORINE).

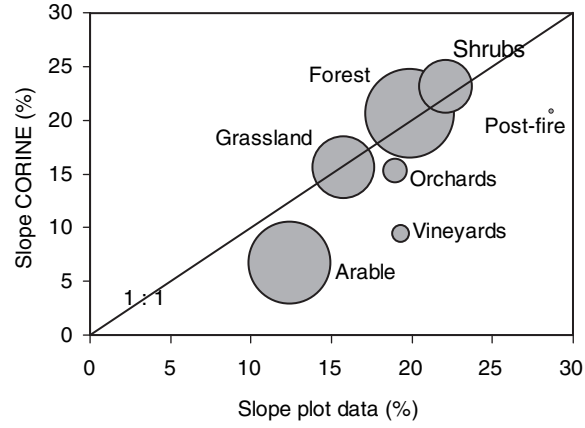

Figure 2.4.5 Mean slope of the reclassified CORINE land covers against the mean slope of the land uses present in the soil erosion database (circle size corresponds to relative area)

2.4.4 CONCLUSION

Sheet and rill erosion rates measured under different climatic, pedological, topographic and land cover conditions have been reviewed using an extensive database of short- to medium-term measurements at the plot scale from western and central Europe. The data confirm the important impact of land use on soil erosion rates. If we rank (in descending order) the land use classes according to the observed sheet and rill erosion rates, we obtain bare soil, vineyard, maize, spring crops, cereal, post-fire, forage, shrubs, grassland and forest. On the basis of these results, we calculated mean sheet and rill erosion rates for the total area in Europe, covered by CORINE ($\sim$1 t ha^{-1} yr^{-1}), and for the erodible areas ($\sim$1.6 t ha^{-1} yr^{-1}). However, from our calculations, it appears that arable land in general and more specifically vineyards ($\sim$20 t ha^{-1} yr^{-1}) and spring crops ($\sim$12 t ha^{-1} yr^{-1}), which represent 16.7 % of the total area covered by CORINE, suffer from significant soil erosion problems. These values should be treated with caution, given the difficulties encountered when extrapolating point measurements to regions. Nonetheless, they are based on measured soil loss values under natural rainfall and indicate an average soil loss below the previous published value of 17 t ha^{-1} yr^{-1} for Europe (e.g. Pimentel *et al.*, 1995).

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2.5

Gully Erosion in Europe

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2.5.1 INTRODUCTION

Soil erosion by water has received and still receives a lot of attention from scientists, soil conservationists and policymakers in Europe. However, when dealing with this soil degradation process, attention has merely been focused on sheet (interrill) and rill erosion rather than on gully erosion. This is reflected in the scientific literature, where more than 2200 plot-year data on soil loss by sheet and rill erosion in Europe have been published (see Chapter 2.4). Such data have been and are still being used to assess the impacts of land use on soil loss by water erosion or to develop, calibrate and validate various empirical and process-based water erosion models (addressing mainly sheet and rill erosion). Such models are then used for assessing soil erosion under global change or for establishing soil erosion risk maps at various scales (e.g. Van der Knijff *et al*., 2000). However, in many European landscapes under different pedo-climatic conditions and with different land uses one can observe the presence and dynamics of various gully types, e.g. ephemeral gullies, permanent or classical gullies and bank gullies. Field-based evidence suggests that sheet and rill erosion as measured on runoff plots are not always realistic indicators of total catchment erosion in Europe, nor do they indicate adequately the redistribution of eroded soil within a field. It is through (ephemeral) gully erosion that a large fraction of soil eroded within a field or catchment is redistributed and delivered to water courses (e.g. Evans, 1993; Martínez-Casasnovas *et al*., 2002).

Gully erosion in Europe has received much less attention than sheet and rill erosion, despite the fact that pictures from large or deep gullies are often shown to illustrate the seriousness of soil erosion by water in

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particular European study areas. Most studies, however, then zoom in on processes of runoff generation, splash erosion, interrill and rill erosion operating within the intergully zones without bothering too much about how these processes relate to gully erosion. Several reasons for the limited attention given to gully erosion can be put forward: (1) runoff plot research and available soil erosion models have focused the attention of people mainly on sheet and rill erosion, (2) gully erosion often occurs at spatial scales, which are beyond the traditional scale for investigating soil erosion by water (e.g. cultivated plots or parcels), (3) as gully erosion often occurs outside the boundaries of a field parcel, gully erosion is often not seen by farmers as a problem of their concern, (4) gully erosion is usually caused by more intense and hence less frequent climatic events and is therefore more difficult to measure and (5) gully erosion is apparently difficult to model.

Recent field-based studies indicate that (1) gully erosion is an important soil degradation process in a range of European environments, causing considerable soil losses and producing large volumes of sediment (e.g. Poesen and Valentin, 2003) and (2) (ephemeral) gully development increases the sediment connectivity in the landscape and hence also the sediment delivery to lowlands and permanent water courses where gullies aggravate off-site effects of water erosion (such as sediment deposition, flooding and pollution; e.g. Poesen and Hooke, 1997; Poesen *et al*., 2003). Many cases of sediment and chemical pollution of watercourses and damage to properties by runoff from agricultural land relate to the occurrence of (ephemeral) gully erosion (e.g. Verstraeten and Poesen, 1999; Boardman, 2001; Ramos and Martínez-Casasnovas, 2004). However, soil losses caused by (ephemeral) gully erosion are rarely accounted for in current soil loss assessment programmes in Europe. For recent literature reviews on gully erosion in general, the reader is referred to Bull and Kirkby (2002) and Poesen *et al*. (2002, 2003).

The objective of this chapter is to provide more insight into the phenomenon of gully erosion in Europe by addressing the following questions.

1. What is gully erosion?
2. Where do gullies typically occur in Europe?
3. Is gully erosion a recent phenomenon in Europe?
4. How important is gully erosion in Europe?
5. What are the major consequences of gully erosion in Europe?
6. What are major triggering and controlling factors of gully erosion?
7. Do we have reliable gully erosion models in Europe?
8. How can gully erosion be prevented or controlled?
9. What are the main research needs for a better understanding of gully erosion and its control?

2.5.2 WHAT IS GULLY EROSION?

Gully erosion is defined as the erosion process whereby runoff water accumulates and often recurs in narrow channels and, over short periods, removes the soil from this narrow area to considerable depths (Figure 2.5.1). Permanent *gullies* (e.g. Figure 2.5.2) are often defined for agricultural land in terms of channels too deep to easily obliterate with ordinary farm tillage equipment, typically ranging from 0.5 to as much as 25–30 m in depth (Soil Science Society of America, 2001).

In the 1980s, the term *ephemeral gully erosion* (e.g. Figure 2.5.1 and 2.5.3) was introduced to include concentrated flow erosion phenomena larger than rill erosion but less than classical gully erosion, as a consequence of the growing concern that this sediment source used to be overlooked in traditional soil erosion assessments (Foster, 1986; Grissinger, 1996a,b). Even though in the literature ephemeral gullies are recorded on many photographs of erosion, it is only during the last two decades that these erosion phenomena have been recognised as being a

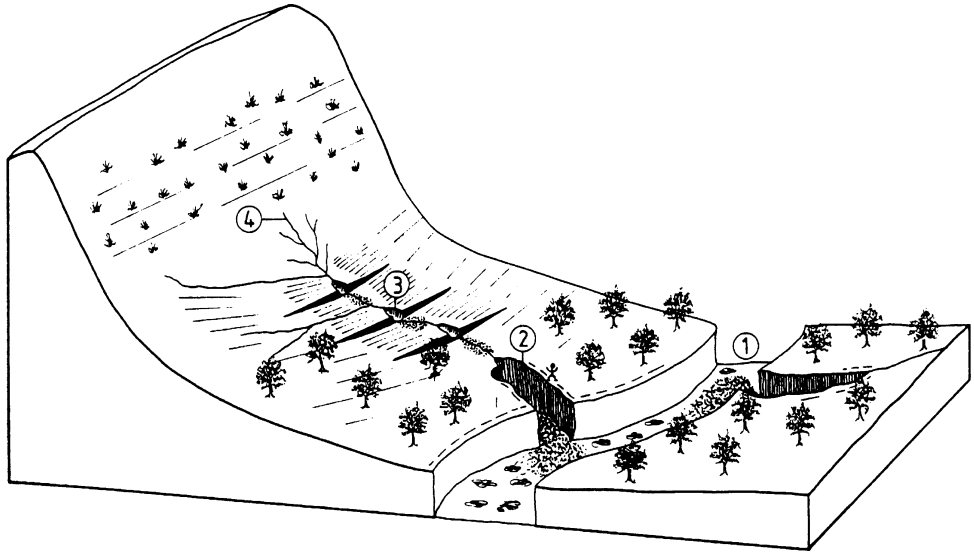

Figure 2.5.1 Sketch of a south European landscape illustrating the typical location of the various gully types discussed in this chapter. 1, River channel; 2, bank gully that developed in a river bank and the gully head that retreated in an orchard; 3, bank gully that developed in a terrace bank; and 4, ephemeral gully in cultivated land or permanent gully in rangeland. [From Poesen J *et al*., Gully erosion in dryland environments, in *Dryland Rivers: Hydrology and Geomorphology of Semi-arid Channels*, Bull LJ, Kirkby MJ (eds), 2002, 229–262. Copyright John Wiley & Sons, Ltd. Reproduced with permission]

Figure 2.5.2 Permanent gully that developed in abandoned cropland (nowadays degraded rangeland) on a pediment (September 1996, El Nazareno, Almeria, Spain)

Figure 2.5.3 Ephemeral gully that developed in a maize field (July 1997, Bertem, central Belgium). Since the soil profile in the concentrated flow zone has locally been truncated, no resistant Bt horizon is present which resulted in a channel incision up to 0.80 m

major part of the erosional systems on cropland (Evans, 1993). According to the Soil Science Society of America (2001), *ephemeral gullies* are small channels eroded by concentrated overland flow that can be easily filled by normal tillage, only to form again in the same location by additional runoff events. Poesen (1993) observed ephemeral gullies to form in concentrated flow zones, located not only in natural drainage lines (thalwegs of zero-order basins or hollows) but also along (or in) linear landscape elements (e.g. drill lines, dead furrows, headlands, parcel borders, access roads). Channel incisions in linear landscape elements are usually classified as rills according to the traditional definitions that associate rill formation with the micro-relief generated by tillage or land forming operations (Haan *et al*., 1994). However, such incisions may also become very large, so this classification seems unsuitable. In order to account for any type of concentrated flow channel that would never develop in a conventional runoff plot (used to measure rates of interrill and rill erosion), Poesen (1993) distinguishes rills from (ephemeral) gullies by a critical cross-sectional area of 929 cm^2 (square foot criterion). Hauge (1977) first used this criterion. Other criteria include a minimum width of 0.3 m and a minimum depth of about 0.6 m (Brice, 1966), or a minimum depth of 0.5 m (Imeson and Kwaad, 1980). As to the upper limit of gullies, no clear-cut definition exists. In other words, the boundary between a large gully and a(n) (ephemeral) river channel is very vague. Nevertheless, it must be acknowledged that the transition from rill erosion over ephemeral gully erosion and classical gully erosion to river channel erosion (Figure 2.5.1) represents a continuum, and any classification of hydraulically related erosion forms into separate classes, such as microrills, rills, megarills, ephemeral gullies and gullies, is, to some extent, subjective (Grissinger, 1996a,b). In fact, Nachtergaele *et al*. (2002a) demonstrated that the relationship between mean width of (ephemeral) gullies and concentrated flow discharge is very similar to the corresponding relation for rills and (small) rivers.

Figure 2.5.4 Bank gullies that developed in an ephemeral channel bank consisting of gravelly sandy loam. Gully headcut retreats into cropland (October 1996, Zarcilla de Ramos, Murcia, Spain)

By definition, *bank gullies*, also termed *edge-of-field gullies* in North America (Dabney *et al*., 2004) (see Figures 2.5.1 and 2.5.4), develop wherever concentrated runoff crosses an earth bank (e.g. river bank, terrace bank, sunken lane bank, lynchet or quarry bank). Given that the local slope gradient of the soil surface at the bank riser is very steep (i.e. subvertical to vertical), bank gullies can rapidly develop at or below the soil surface by hydraulic erosion, piping and eventually mass movement processes even though catchment areas are very small (Poesen and Govers, 1990). Once initiated, bank gullies retreat by headcut migration into the more gentle sloping soil surface of the bank shoulder and further into low-angled pediments, river or agricultural terraces (Poesen *et al*., 2002).

2.5.3 WHERE DO GULLIES TYPICALLY OCCUR IN EUROPE?

Gully erosion occurs throughout Europe. Typically, gullies can be found in croplands, rangelands and badlands. Figure 2.5.5 indicates on a map of western Europe (compiled by De Ploey, 1989a, based on literature and field observations) areas with arable land where rates of soil erosion by water may regularly exceed $10\,\mathrm{t\,ha^{-1}\,yr^{-1}}$. Such areas are characterised by periods during the year with low or negligible vegetation cover. If, in addition, the soils in these areas are susceptible to surface sealing, significant volumes of Hortonian runoff can be produced during rainfall, which in concentrated flow zones with slope gradients in excess of 4–5 % leads to the development of ephemeral gullies (e.g. Figure 2.5.3) and bank gullies (Figure 2.5.4).

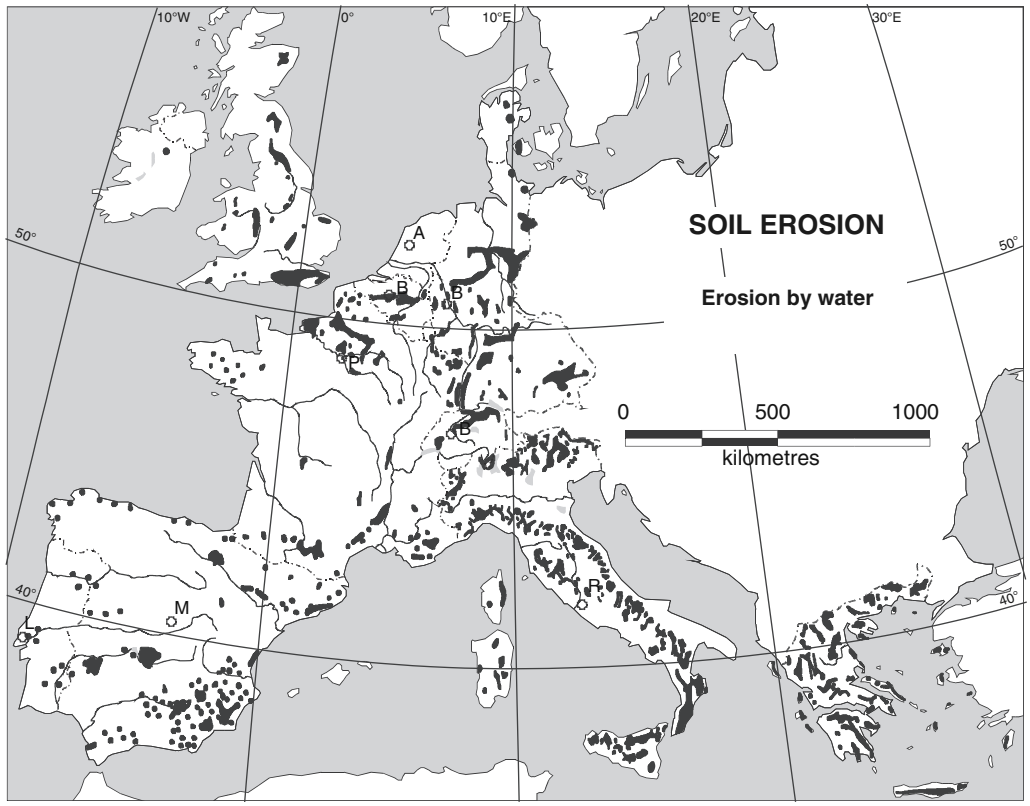

Figure 2.5.5 Map of western Europe indicating regions where soil erosion by water regularly exceeds soil loss tolerance levels. In most of these regions cropland is the dominant land use and ephemeral gully erosion frequently occurs. (From De Ploey J, *Soil Erosion Map of Europe*, 1989. Reproduced by permission of Catena-Verlag GmbH)

Active permanent gullies can be observed in a wide range of environments, from degraded rangelands (due to overgrazing or after burning schrubs), both in northern Europe (e.g. Iceland) and in Mediterranean Europe, to peatland (e.g. Wishart and Warburton, 2001).

Spectacular permanent, dense gully networks can be found in badland areas in the Mediterranean (Figure 2.5.6 and 2.5.7) but also in other severely eroded areas such as in Iceland. Badlands result from both water erosion and mass movement processes. These processes interact and their effects are therefore difficult to separate from each other, e.g. gullying by hydraulic erosion followed by gully wall collapse (mass failure) (Poesen and Hooke, 1997). Lithological conditions are important and badlands tend to develop on unconsolidated or poorly sorted materials such as shales, gypsiferous and salty-silty marls and silt-clay deposits of Tertiary and Quaternary age. Most badlands are situated on or near major mountain ranges, especially on those that are still being uplifted. Badlands evolve by surface and subsurface erosion by water, including chemical erosion (soil dispersion due to the high concentration of salts) and piping (Torri *et al*., 2000; Gallart *et al*., 2002). Characteristics of active badlands are high contemporary erosion rates, low surface permeabilities and high erodibilities. Measured erosion rates in Mediterranean badlands vary widely, ranging between 5 and 220–330 t ha^{-1} yr^{-1} (e.g. Benito *et al*., 1992; Bufalo and Nahon, 1992; Martínez-Casasnovas and Poch, 1998). This wide range is the result of differences in climatic, lithologic and topographic characteristics at the various study sites, differences in spatial and temporal scales considered, and also differences in measurement and calculation techniques used in the various studies.

Figure 2.5.6 Dense gully network in badlands that developed in marls (April 1999, Librilla, Murcia, Spain). The presence of such a gully network near a reservoir increases the runoff and sediment connectivity significantly, leading to rapid siltation of the reservoir

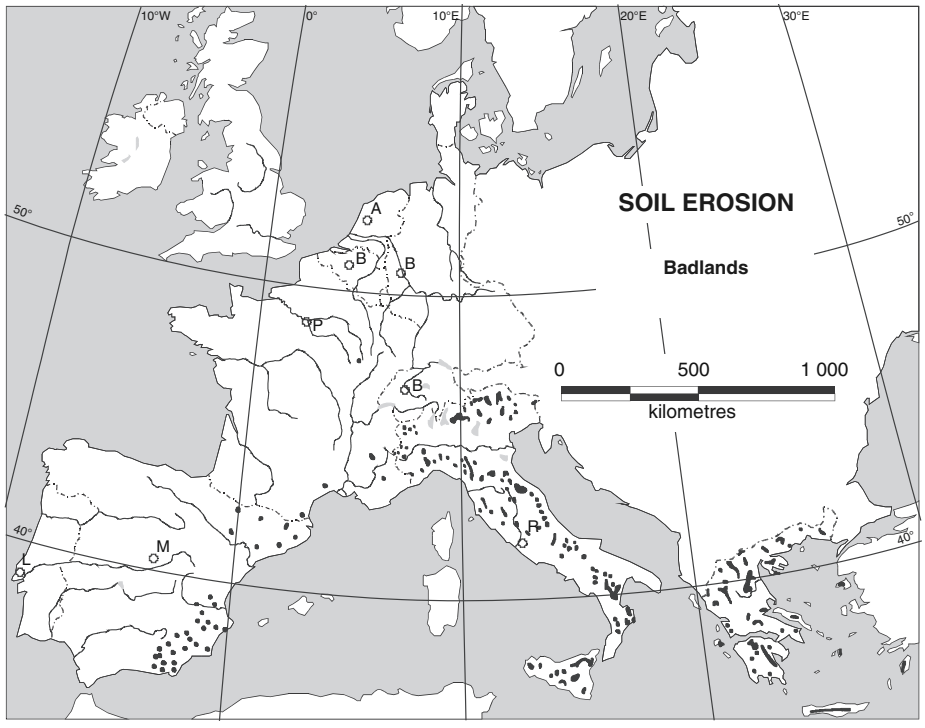

Figure 2.5.7 Map of western Europe indicating regions with dense gully networks in badlands. (From De Ploey J, *Soil Erosion Map of Europe*, 1989. Reproduced by permission of Catena-Verlag GmbH)

2.5.4 IS GULLY EROSION A RECENT PHENOMENON IN EUROPE?

There are several reports indicating that throughout Europe gully development has been locally significant over the last 3000 years. Some of these gullies are still clearly visible, for instance in forested areas of Belgium (Gullentops, 1992; Vanwalleghem *et al*., 2003, 2005c), France (Vogt, 1953), Germany (Semmel, 1995; Bork *et al*., 1998), Slovakia (Stankoviansky, 2003) and Hungary (Gábris *et al*., 2003), where the forests have preserved these geomorphic phenomena, or in stabilized badlands of the Mediterranean (e.g. Wise *et al*., 1982).

In northern Europe, for instance, recent reports indicate that gully erosion occurred as early as the late Neolithic (2857–2495 BC) in Germany (Schmidtchen and Bork, 2003), the late Bronze Age (ca 1700 BC) and the end of the Roman period in central Belgium (Vanwalleghem *et al*., 2005d), in the 9–10th centuries in the UK (Harvey, 1996), in the 14th century in Germany (Bork *et al*., 1998; Dotterweich *et al*., 2003) and Slovakia (Stankoviansky, 2003) or during the Little Ice Age in Belgium (Vanwalleghem *et al*., 2005b), Germany (Bork *et al*., 1998; Dotterweich *et al*., 2003) and in Slovakia (Stankoviansky, 2003). For Mediterranean Europe, various studies reported that gully erosion already occurred in prehistoric times (e.g. Wainwright and Thornes, 2004). For instance, Allée and Denèfle (1989) report that gullying has been initiated in the eastern French Pyrenees from ca 650 BC on. De Ploey (1992) calculated the age of badlands in the Mediterranean to range between 2700 and 40 000 years. It is very likely that throughout historical times, gully erosion was significant in the Mediterranean. For instance, Vandekerckhove *et al*. (2001) calculated the age of active bank gullies in southeast Spain and found that they initiated in a time span ranging between 350 and 1940 AD. Torri and Rodolfi (2000) report that badlands in central Italy were initiated around 1850.

In conclusion, gully erosion in Europe is not a recent phenomenon. Several studies report that gully development coincided with periods of land clearing, often in combination with very intense rains resulting in a change of catchment hydrology in response to changing environmental conditions. From a detailed study of an infilled gully under cropland in Belgium, Vanwalleghem *et al*. (2005b) concluded that over the last 350 years, at least five cut and fill cycles occurred, indicating that the landscape reacts in a very dynamic way to gully incision. Under cropland, an ephemeral gully can develop into a large permanent gully over a few months or years, but within subsequent decades, the entire gully can be almost completely filled in again if there is continuous cultivation in the catchment with runoff and sediment production. Much can be learned from detailed studies on environmental change leading to intensive gullying (Poesen *et al*., 2003).

2.5.5 HOW IMPORTANT IS GULLY EROSION IN EUROPE?

Data on gully erosion rates reported in the literature have been compiled in Table 2.5.1. In European cropland, mean rates of ephemeral gully erosion range between 1 and 40 $t\,ha^{-1}\,yr^{-1}$ depending on rainfall and site conditions (Table 2.5.1). However, the highest soil erosion rates in Europe have been recorded in active badland areas where gully erosion is the dominant erosion process. In such areas, soil losses at the catchment scale equal 57–137 $t\,ha^{-1}\,yr^{-1}$ in badlands of the Alpes de Haute Provence (France; Mathys *et al*., 2003), 123 $t\,ha^{-1}\,yr^{-1}$ in the Pinedes region (northeastern Spain; Martínez-Casasnovas *et al*., 2003), 190 $t\,ha^{-1}\,yr^{-1}$ in densely gullied badlands on black marls in southeastern France (Bufalo and Nahon, 1992) and even 302–455 $t\,ha^{-1}\,yr^{-1}$ in badlands located within the basin of the Barasonas reservoir in north-eastern Spain (Martínez-Casasnovas *et al*. 2003) (Table 2.5.1).

The contribution of gully erosion to total soil loss by water erosion in Europe is variable and ranges between 10 and 83 % (Table 2.5.1). Factors controlling this contribution are size of the study area, time-

TABLE 2.5.1 Soil loss rates due to gully erosion (SLgully) and contribution of (ephemeral) gully erosion to overall soil loss rates and to sediment production rates by water erosion: SLgully (%) = 100 × (ratio between SLgully and total SL rates due to interrill, rill and gully erosion)

Location	SLgully ($t\,ha^{-1}\,yr^{-1}$)	SLgully (%)	Source
Belgium, central	22.3	10	Govers and Poesen (1988)
Belgium	1.1–5.9	n.a.[a]	Nachtergaele and Poesen (1999)
France, north	n.a.	10–45	Ludwig *et al*. (1992)
Germany, south	n.a.	12–29	Auerswald (1998)
France, Normandy	n.a.	21–56	Cerdan *et al*. (2002)
France, south-east	190	n.a.	Bufalo and Nahon (1992)
Spain, north-west	1.5	26	Valcarcel *et al*. (2003)
Germany, south-west	n.a.	36	Baade (1994)
Romania	n.a.	37	Nedelcu (1999)
Belgium, central	3.6	44	Poesen *et al*. (1996)
France, north	n.a.	46–55	Auzet *et al*. (1995)
Italy, Sicily	5.0	n.a.	Capra and Scicolone (2002)
Portugal, Bragança	16.1	47	Vandekerckhove *et al*. (1998)
Spain, Guadalentin	37.6	51	Poesen *et al*. (2002)
Norway, Leira basin	12.7	55	Bogen *et al*. (1994)
Spain, Catalunia	n.a.	58	Martínez-Casasnovas *et al*. (2002)
Spain, Catalunia	123	n.a.	Martínez-Casasnovas *et al*. (2003)
Spain, Northeast	302–455	n.a.	Martínez-Casasnovas *et al*. (2003)
Spain, south-east	1.2	59	Oostwoud Wijdenes *et al*. (2000)
Belgium, central	n.a.	60	Quine *et al*. (1994)
Spain, north	64.9	74	Casali *et al*. (2000)
Portugal, Alentejo	3.2	80	Poesen *et al*. (1996)
Spain, Almeria	9.7	83	Poesen *et al*. (1996)

[a]Data not available.

scale considered, climate and magnitude of rain event(s), topography, soil type and land use (Poesen *et al*., 2003).

Few studies have reported rates of gully expansion in Europe. Oostwoud Wijdenes *et al*. (2000) observed that land use has a significant impact on the expansion of bank gully heads in southeastern Spain. Reported mean linear headcut retreat rates range between $0.1\,m\,yr^{-1}$ (min. = 0.01; max. = $0.62\,m\,yr^{-1}$) for active gully headcuts in southeastern Spain (Vandekerckhove *et al*., 2001) and $0.92\,m\,yr^{-1}$ (min. = 0.42; max. = $1.83\,m\,yr^{-1}$) in the Moldavian Plateau of eastern Romania (Ionita, 2000). For large gullies in the Penedes region (northeastern Spain), Martínez-Casasnovas (2003) measured an average rate of gully wall retreat of $0.2\,m\,yr^{-1}$, with maximum rates of channel expansion of 0.7–$0.8\,m\,yr^{-1}$, occurring at the gully head and at meandering gully bends. Vandekerckhove *et al*. (2003) reported medium-term (40–43 years) mean volumetric headcut retreat rates of active gullies in southeastern Spain of $17.4\,m^3\,yr^{-1}$. Differences in retreat rates between gullies could be largely explained by drainage areas. Nachtergaele *et al*. (2002b) monitored the length, surface area and volume of a gully developing in a loess-derived soil (central Belgium) over a 13-year period and reported a degressive increase of gully extension over time which could be largely explained by changing topographic variables at the gully head (i.e. slope gradient and drainage area). Similar observations for gullies in Romania were reported by Radoane *et al*. (1999). Gullies not only expand by headcut retreat, but also by channel widening. Martínez-Casasnovas *et al*. (2004) assessed

sidewall erosion in large gullies in northeastern Spain and reported soil losses for two successive periods of 16 and 83 t ha^{-1} yr^{-1}, the latter depending on the occurrence of an extreme rain event during the observation period.

From the data reported in this section, we conclude that soil loss rates caused by gully erosion are far from negligible and that they can exceed soil loss rates for other water erosion processes.

2.5.6 WHAT ARE THE MAJOR CONSEQUENCES OF GULLY EROSION IN EUROPE?

Here, we discuss briefly the major on- and off-site consequences of gully erosion.

2.5.6.1 On-site

The most obvious and important on-site consequence of gully erosion is the loss of soil, which can be of the same order of magnitude as soil losses due to sheet and rill erosion and in ca 50 % of cases even more (see Table 2.5.1). Hence, gully erosion is a significant soil degradation process.

Figure 2.5.8, which is based on experimental data collected in three European study areas, shows how gully length, gully area and gully volume evolve over time. From these graphs, it is clear that gully erosion increases over time but in a degressive manner. In other words, 50 % of total gully length, total gully area or total gully volume is created in 20 % or less of the total gully lifetime. Hence, if gullies are allowed to evolve without direct interference by human activities (e.g. infilling, land levelling, ploughing), gully erosion rates (caused by headcut and bank retreat) usually slow over time. However, in cropland, ephemeral gullies are typically filled in by tillage (soil translocation by tillage leading to tillage erosion and tillage deposition; see Chapter 2.9) within less than 1 year after their development. During subsequent storms, the infilled loose soil is usually eroded again by concentrated flow, thereby increasing the plan-form concavity of the site. The newly created plan-form concavity increases the probability of erosive concentrated flow (Poesen *et al*., 2003). Hence ephemeral gully erosion and tillage erosion reinforce each other.

In various parts of Europe, landscapes heavily dissected by gullying (badlands) have been levelled (e.g. Norway, Mediterranean countries), thereby causing strong soil profile truncation in the intergully areas and infilling of gullies with the translocated soil material (e.g. Poesen and Hooke, 1997; see Chapter 2.12). Such land levelling operations have often resulted in renewed gully incision of the levelled land and also in shallow landsliding, causing very large soil loss rates. In other words, important interactions exist between gully erosion on the one hand and tillage and land levelling operations on the other (Poesen *et al*., 2003).

The gully channels that develop change the local topography drastically and cause a decrease in several soil functions (e.g. bearing function, archive function, plant-growth function). Furthermore, these channels render trafficability very difficult or almost impossible.

Once gully channels have developed, the water infiltration rate through the gully bottom may be significantly larger than that of the soil surface in the intergully areas if the gully channel develops into more permeable soil horizons (Poesen *et al*., 2003). As such, gully channels may contribute to runoff water transmission losses. Where gully channels are used as garbage dumps, as is often the case, this may lead to significant groundwater contamination. However, if gullies develop into hillslopes with temporary water tables, gully channel development may cause enhanced drainage of the hillslope and a rapid water-table lowering, which leads to a drying out of the soil profiles in the intergully areas (Poesen *et al*., 2003). As a consequence, in dry Mediterranean areas crop production in the intergully area may be adversely affected by gullying.

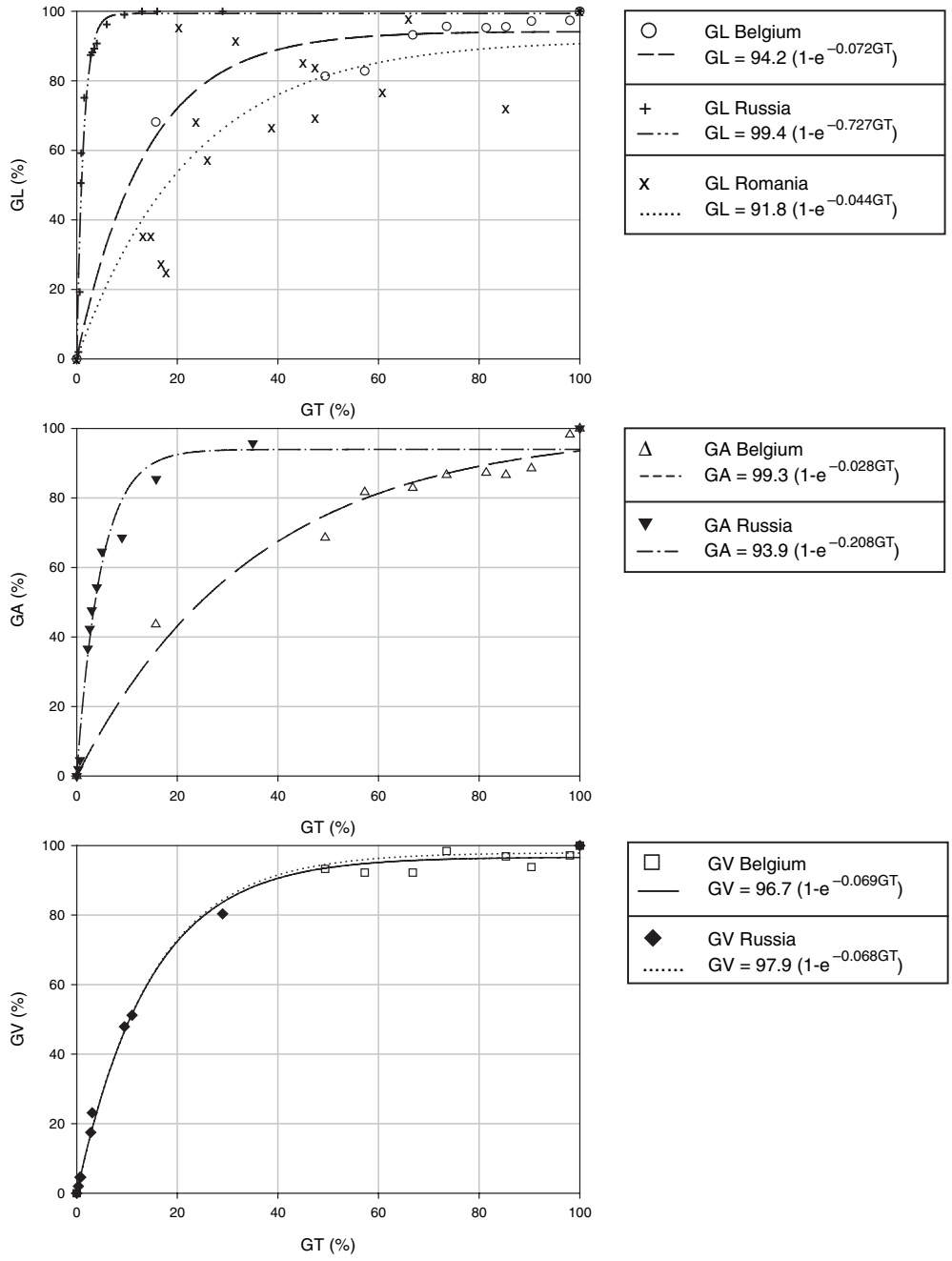

Figure 2.5.8 Evolution of gully length (GL, %), gully surface area (GA, %) and gully volume (GV, %) during gully lifetime (GT, %) for different study areas (partly based on Vanwalleghem *et al*., 2005c). All parameters are given as percentages, relative to the last measured parameter value. Data for Belgium are given by Nachtergaele *et al*. (2002b) and Vanwalleghem *et al*. (2005b) and are based on field measurement of the evolution of a permanent gully in loess. Gully lifetime (GT) = 14.8 years. Data for Russia are given by Kosov *et al*. (1978) and are based on laboratory experiments of gully formation in sand. The data points are extrapolated from a graph. Gully lifetime (GT) is unknown. Data for Romania are given by Surdeanu *et al*. (2003) and are based on a map survey of gully evolution in marly clay rocks. The data points are extracted from a scatter plot. Gully lifetime (GT) = 325 years

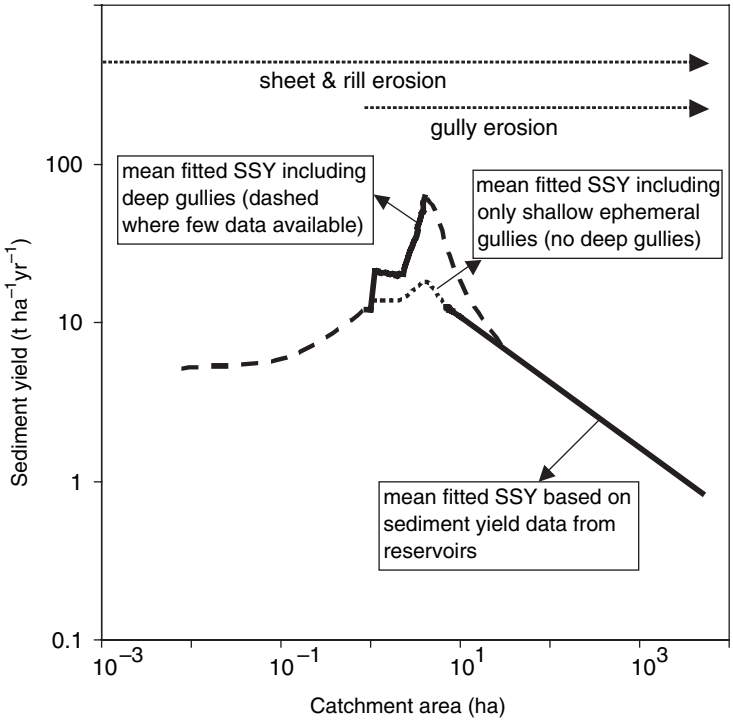

Figure 2.5.9 Relationship between catchment area and area-specific sediment yield (SSY) for cropland in the loess belt. Trends in SSY are fitted through soil loss data collected using runoff plots, volumetric measurements of shallow and deep ephemeral gullies and sediment deposition volumes in flood retention ponds, all located in central Belgium. (Reprinted from Vanwalleghem T *et al*., Characteristics, controlling factors and importance of deep gullies under cropland on loess-derived soils, *Geomorphology*, **69**: 76–91, 2005, with permission from Elsevier)

2.5.6.2 Off-site

Gully erosion represents a very significant sediment source. This is well illustrated in Figures 2.5.9 and 2.5.10. Sediment yield (SSY) clearly depends on the size of the catchment for which the data have been collected. In the Belgian loess belt, for instance, the mean SSY for areas less than 1 ha is usually less than 10 t ha^{-1} yr^{-1} (Figure 2.5.9). For areas between 1 and 10 ha, however, SSY increases to 20 t ha^{-1} yr^{-1} if shallow ephemeral gullies develop, and may rise to 60 t ha^{-1} yr^{-1} if deep (>0.8 m) gully channels develop (Vanwalleghem *et al*., 2005a). For catchments larger than 10 ha, SSY decreases because of an increased probability of sediment deposition taking place. A similar trend, although with a peak SSY occurring between 10 and 1000 ha, has been reported for Spain (de Vente and Poesen, 2005; Figure 2.5.10). In both case-study areas, the development of gullies in areas larger than 1 ha may be held responsible for the rapid increase in sediment yield with increasing drainage area.

Not only do expanding gullies produce large volumes of sediment, but gullies also form effective links in the landscape, transferring both runoff and sediment (produced in the intergully areas) from uplands to valley bottoms. In other words, gullies increase the connectivity for sediment in the landscape (Poesen *et al*., 2003). As a consequence, gully erosion contributes significantly to reservoir siltation, as illustrated by Figures 2.5.6 and 2.5.11 (see also Chapter 2.20), and muddy floods (see Chapter 2.19). Active gullies are thus important indicators for high sediment production rates within catchments (Poesen *et al*., 2003; Verstraeten *et al*., 2003; de Vente *et al*., 2005).

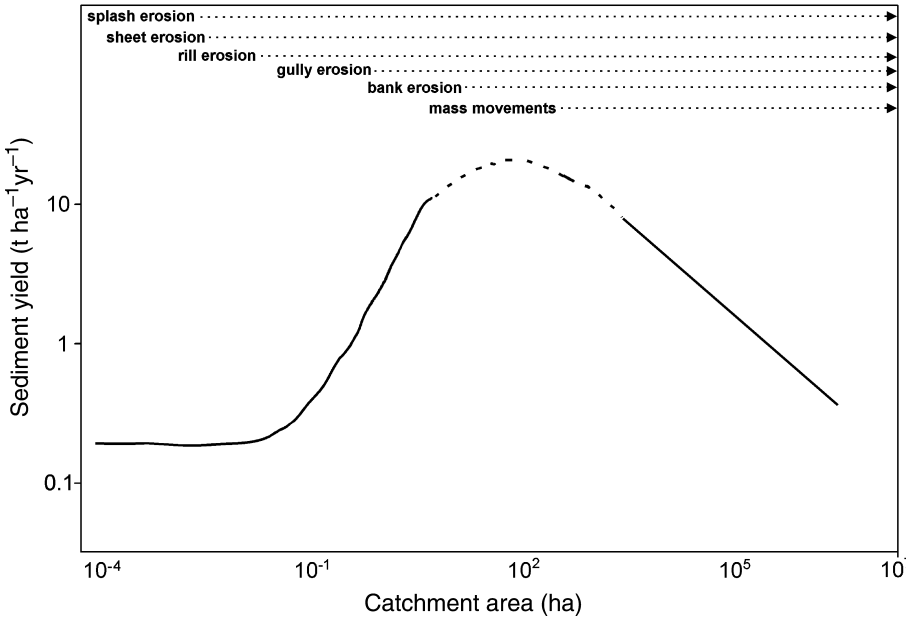

Figure 2.5.10 Relationship between catchment area and area-specific sediment yield (SSY) for Spain. Trends in SSY are fitted through soil loss data collected using runoff plots, volumetric measurements of gullies and sediment deposition volumes in reservoirs, all located in Spain. Dominant soil erosion process for each spatial scale is indicated as well. (Reprinted from de Vente J and Poesen J, Predicting soil erosion and sediment yield at the basin scale: scale issues and semi-quantitative models, *Earth Science Reviews*, **71**: 91–125, 2005, with permission from Elsevier)

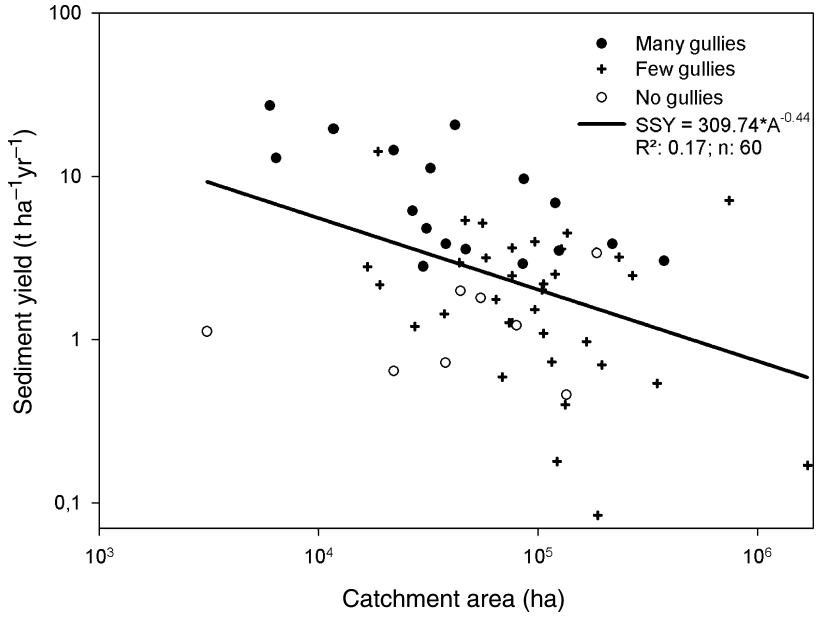

Figure 2.5.11 Relationship between catchment area, area-specific sediment yield (based on reservoir sedimentation data reported by Avendaño Salas *et al.*, 1997) and the presence of gullies. (Reprinted from de Vente J *et al.*, The application of semi-quantitative methods and reservoir sedimentation rates for the prediction of basin sediment yield in Spain, *Journal of Hydrology*, **305**: 63–86, 2005, with permission from Elsevier)

2.5.7 WHAT ARE MAJOR TRIGGERING AND CONTROLLING FACTORS FOR GULLY EROSION?

The probability of gullies developing during concentrated overland flow increases if several conditions are met. First, during a rain event, concentrated flow intensity (usually expressed by boundary flow shear stress $\tau_b = \rho g d s$, where ρ = density of runoff water, g = acceleration due to gravity, d = depth of flow and s = sine of the soil surface gradient) must exceed a critical value (in this case the critical flow shear stress, τ_c), so that a channel exceeding the critical cross-section (i.e. 929 cm^2) can be cut in the topsoil. Reported τ_c values for incipient gully channel development on cropland typically range between 5 and 90 Pa but can be as high as 240–260 Pa for well-established pasture (Poesen *et al*., 2003). Second, a large channel erodibility (*Kc*) value of the soil in which a gully might develop increases the probability of gully incision during a rain storm.

Various factors affect both τ_b and *Kc*. Rain depth and intensity, topography [slope gradient (*S*), drainage area (*A*), plan concavity], soil type and land use affect τ_b whereas soil type and landuse affect *Kc*. Critical rain event depths for the initiation of ephemeral gullies in European cropland range between 15 and 22 mm (Poesen *et al*., 2003). On cropland with poor vegetation cover, and hence a significant Hortonian runoff production, ephemeral gullies usually start to develop in concentrated flow zones when local slope gradients exceed 3–4 %. Critical *S*–*A* relations for gully initiation and for sediment deposition (at the bottom end of gully channels) in a range of European environments have been reported by Vandaele *et al*. (1996), Vandekerckhove *et al*. (2000), Nachtergaele *et al*. (2001a,b) and Poesen *et al*. (2003). Soils prone to (ephemeral) gully development in Europe are soils that developed on loess, sandy loams, marls and volcanic ashes. In addition to soil type, the vertical distribution of erosion resistance of the various soil horizons largely controls the depth, the cross-sectional morphology and hence the total eroded soil volume by gullies. For instance, the presence of a well-developed Bt horizon in loess-derived soils drastically reduces gully depth in the loess belt. Where the Bt horizon has been eroded, deeper gullies can develop (Poesen, 1993; Nachtergaele and Poesen, 2002, Vanwalleghem *et al*., 2005a). In Mediterranean environments, the presence of very stony soils with hard unweathered bedrock at shallow depth in the soil profile (Leptosols) limits the development of deep gullies. Land use significantly affects gully erosion rates. On poorly vegetated areas (cropland, degraded rangeland), large volumes of Hortonian runoff are usually produced, resulting in large shear stresses in concentrated flow zones. If, in addition, the land is tilled, channel erodibility (*Kc*) is fairly high as illustrated in Figure 2.5.12b (after Knapen *et al*., submitted). If the land is left untilled, the mean *Kc* is on average a factor 10 smaller, resulting in a much smaller probability of gully development. Tillage results in a loosening of the topsoil and therefore in a reduction in soil cohesion and a drastic increase in channel erodibility. In general, the transformation of natural vegetated slopes to tilled cropland or very degraded rangeland causes a significant lowering of the topographic threshold for gully development (Poesen *et al*., 2003). This scenario explains in many cases the presence of old gullies on slopes which were formerly cultivated but are nowadays covered by natural vegetation. Along the same lines, Oostwoud Wijdenes *et al*. (2000) reported that the shift from matorral to cropland (i.e. wheat and intensively cultivated almond groves) in Spain resulted in a drastic reactivation of bank gullies. Not only the aboveground biomass but also the roots play an important role in increasing the soils'resistance to concentrated flow erosion (Gyssels *et al*., 2005).

2.5.8 DO WE HAVE RELIABLE GULLY EROSION MODELS IN EUROPE?

In contrast to sheet and rill erosion models, few models have the potential to predict gully erosion rates. One of these is the EGEM (Ephemeral Gully Erosion Model; Merkel *et al*., 1988; Woodward, 1999), developed in the USA. This model was tested for its suitability to predict ephemeral gully erosion rates in various European cropland environments (i.e. Belgium, Portugal, Spain and Italy) (Nachtergaele *et al*., 2001a,b; Capra *et al*., 2005).

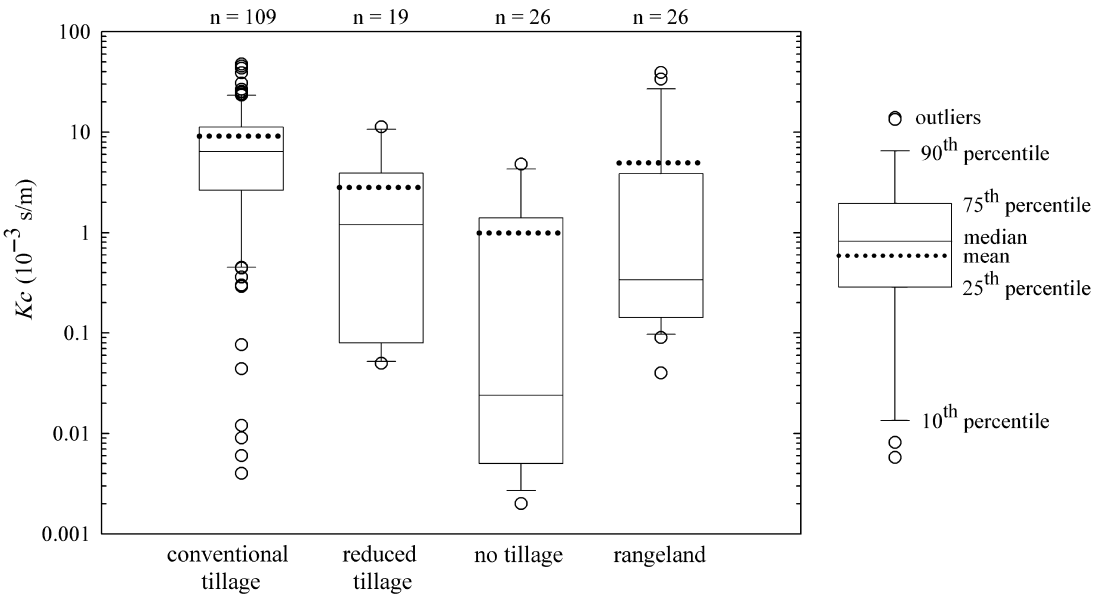

Figure 2.5.12 Channel erodibility (*Kc*) measured during concentrated flow erosion experiments on soils having different textures under different land use or tillage treatments. Published experimental data come from all over the world; *n* is total number of tested soils in each land use class. (After Knapen *et al*., submitted)

From these studies, it can be concluded that EGEM is not capable of predicting ephemeral gully erosion properly in the studied cropland environments. These studies also point to the fact that the ephemeral gully length (*L*) is a key parameter in determining ephemeral gully volume (Poesen *et al*., 2003): *L* explains in all studies more than 64 % of the variance in ephemeral gully volume.

However, few erosion models have the capacity to predict the exact location of gully initiation points, points where gullies end (i.e. where sediment deposition occurs) and therefore gully lengths. Desmet *et al*. (1999) investigated the possibility of predicting the location of ephemeral gully channels in the loess belt using an inverse relationship between local slope gradient (*S*) and upslope contributing area per unit length of contour (*As*). Along the same lines, Jetten *et al*. (in press) used empirical equations predicting topographic thresholds for gully trajectories in terms of *S* and *As* in order to select critical areas in the landscape where ephemeral gully incision might take place. For such areas, then, the event-based spatially distributed LISEM model was used to predict eroded gully channel dimensions. Kirkby *et al*. (2003) presented power law equations describing the locations of ephemeral and permanent gully channel heads. Souchère *et al*. (2003) combined an expert-based approach and field data to predict the location and volume of ephemeral gullies within the main runoff collector network of agricultural catchments in France. Casali *et al*. (2003) and Torri and Borselli (2003) developed gully cross-section models to predict changes in gully channel width and depth during concentrated flow events in Spain and Italy. However, these models still need calibration.

Once initiated, bank gullies essentially expand by gully headcut retreat and, to a lesser extent by gully wall retreat. Whether a headcut retreats as a single headcut or by multiple headcuts is controlled by factors such as topography, soil type and land use. Several studies have attempted to predict gully headcut retreat in a range of European environments. De Ploey (1989b) developed a process-based headcut retreat model. Radoane *et al*. (1995) reported several regression models linking gully headcut retreat rates in Romania to lithology, drainage basin area, gully length, relief energy of the basin and drainage basin inclination.

Vandekerckhove *et al*. (2001, 2003) proposed several empirical equations allowing one to predict short- and medium-term retreat rates of active gully headcuts in Spain. Although few attempts have been made to develop models for predicting either gully subprocesses or gully erosion in a range of environments, there are still no reliable (validated) models available that allow one to predict impacts of environmental change on gully erosion rates at various temporal or spatial scales in Europe, nor the impacts of gully erosion on sediment yield, hydrological processes and landscape evolution. This lack of knowledge explains why gully erosion is not included in most soil erosion assessments in Europe.

2.5.9 HOW CAN GULLY EROSION BE PREVENTED OR CONTROLLED?

In order to prevent (ephemeral) gullies from developing in cropland, all possible measures leading to an increase in rain infiltration, to a reduction in Hortonian overland flow discharge and hence also to a reduction of flow shear stress (τ_b) need to be applied. At the same time, all measures leading to an increase in erosion resistance of the concentrated flow zone will also reduce the risk of gully development.

Where possible, natural vegetation with well-developed root mats should be established in concentrated flow zones with a high risk of gullying. As a consequence, soil loss and sediment production will be reduced and the connectivity for sediment in the landscape will be interrupted, resulting in a smaller sediment delivery to valley bottoms or river channels (Poesen *et al*., 2003). Often, this approach is not feasible because of interference with other land use and therefore solutions adapted to local agricultural practices need to be found.

One of these solutions is the establishment of grassed waterways (e.g. Ouvry, 1989; Baade *et al*., 1993; Fiener and Auerswald, 2003). Grassed waterways are broad, shallow channels often located within large fields, with the primary function to drain surface runoff from cropland without gullying in the thalweg. To serve this function as effectively as possible, selected fast-growing grasses are sown in the waterway and, once established, the grass is frequently mowed to reduce hydraulic roughness. Whereas grassed waterways are a common (ephemeral) gully erosion control practice in North America, this measure is rarely adopted by farmers cultivating relatively small field plots in Europe. Several studies have come up with alternatives to control ephemeral gully erosion, i.e. conservation tillage, topsoil compaction and double drilling.

Figure 4.5.12 clearly illustrates that conservation tillage practices such as reduced tillage or no tillage lower the channel erodibilty (*Kc*) significantly. Whereas conventional tillage (i.e. mouldboard ploughing) results in a loose, less cohesive and hence more erodible plough layer that is easily eroded by concentrated flow, the application of no tillage in the concentrated flow zones will increase the topsoil resistance, as observed in France, Belgium and Italy (e.g. Ouvry, 1989; Poesen and Govers, 1990; Ludwig and Boiffin, 1994; Poesen *et al*., 2003). However, Ludwig and Boiffin (1994) reported that the effects of no tillage on ephemeral gully erosion rates in France largely depend on the spatial location of the no-till treated plots within the catchment and that no tillage was overall less effective than grassed waterways.

As compact and hence more cohesive topsoils or soil horizons have a larger resistance to incision by erosive concentrated flow compared with tilled ones (e.g. Figure 2.5.12), Ouvry (1989) compacted mechanically concentrated flow zones after drilling in France. He observed that this treatment significantly reduced ephemeral gully development within drainage basin areas smaller than 50 ha. Poesen (1993) and Nachtergaele and Poesen (2002) reported that information on the thickness and resistance properties of compact soil horizons (e.g. Bt horizon) is crucial for selecting appropriate soil conservation measures in concentrated flow zones and that any tillage operation (such as subsoiling) resulting in a loosening of compact horizons should be avoided to prevent deep incisons by concentrated flow erosion.

Gyssels *et al*. (2002, personal communication) observed that double drilling of wheat only in concentrated flow zones reduced ephemeral gully erosion rates on average by 25 %, but the reduction could be as high as

50 % in some cases. The effect of double drilling on channel development was particularly clear in the early growth stages of the wheat seedlings because of larger root densities and therefore larger cohesion of the topsoils compared with conventionally drilled topsoils (Gyssels *et al*., 2005).

Once formed, gullies in Europe are usually controlled by check dams. Although check dams in gullies are widespread in Mediterranean Europe, little information is available on their effectiveness and efficiency. For instance, check dams in gullies induce sediment deposition upstream and therefore cause more runoff water with a reduced sediment load to flow downstream of the dam. In many cases, this causes a clear water effect, sometimes resulting in significant channel incision (Boix Fayos *et al*., 2005; Castillo *et al*., 2005). An alternative approach is to establish vegetation barriers on the gully bottom, as documented by Rey (2004).

Despite the several case studies reported in the literature, there is still a need for more information on the effectiveness and cost-efficiency of gully prevention and control measures. Handbooks usually provide the principles on how to control gully erosion, but when applying them in a given environment these techniques often need to be adjusted to local conditions. For instance, Poesen (1989) reported that stabilising a gully headcut in central Belgium with a rock plug did not work in loess-derived soils because of their very high erodibility. In such cases, the use of geomembranes was an effective and efficient alternative.

2.5.10 CONCLUSIONS AND RESEARCH NEEDS

Over the last decade, gully erosion research has contributed significantly to a better understanding of spatial and temporal patterns of gully erosion rates and of controlling factors in Europe. However, several aspects of gully erosion still remain under-researched (Poesen *et al*., 2003):

- Conditions for the initiation, development and infilling of gully channels under a range of environmental conditions;
- Rates and factors controlling gully subprocesses, such as tension crack development, piping, plunge-pool erosion, fluting, bifurcation, mass wasting processes on gully walls and their interactions (e.g. hydraulic erosion and mass wasting processes);
- Models predicting the location of gully channels in the landscape and gully expansion or contraction at different temporal scales;
- Appropriate and standardised monitoring techniques enabling the study of gully development with a higher precision than that obtained by current techniques;
- Detailed monitoring, experimental and modelling work to increase the capacity to predict impacts of environmental changes on gully erosion rates;
- Interaction between gully development, hydrological and other soil degradation processes;
- Innovation in gully erosion control techniques, which is rather limited compared with innovation in gully erosion process research. What can be learned from failures and successes of gully erosion control techniques? What are effective and efficient gully prevention and gully control measures?

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2.6

Piping Hazard on Collapsible and Dispersive Soils in Europe

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2.6.1 INTRODUCTION

Despite early pioneering work on piping (Fletcher *et al*., 1954; Müller-Miny, 1954; Panicucci, 1972; Schröder, 1973; Lulli, 1974; Heede, 1971, 1977; Crouch, 1976; Barendregt and Ongley, 1977), gully erosion research in Europe prior to 1980 was traditionally focused on a Hortonian infiltration-excess model of runoff generation. However, awareness of the importance of piping in erosion in Europe accelerated in the 1980s (Imeson and Kwaad, 1980; Jones, 1981; Alexander, 1982; Bryan and Yair, 1982; Harvey, 1982; Imeson, 1983; Baillie *et al*., 1986; and Gerits *et al*., 1987). This period culminated in a review of gully initiation research by Bocco (1991), who noted that subsurface erosion, particularly piping was an important factor on 60% of the European field cases of gullying that he reviewed. At about this time, Parker and Higgins (1990), Benito *et al*. (1993), Naidu *et al*. (1995) and Sumner and Stewart (1992) were drawing attention to the overwhelming importance of material properties in soil loss and erosion, particularly from case studies in Spain and Australia. Piping-origin rill and gully development is now accepted as a critically important soil erosion process in a wide range of European environments.

The importance stems from the observation that in contrast to soil erosion (rilling and gullying) caused by surface wash, gullies that initiate by the subsurface enlargement of pipes can be responsible for exceptionally high soil losses (Bocco, 1991). This is because by being partly hidden, sometimes beneath well-vegetated slopes, the process can be well advanced before roof collapse reveals the size of the excavated pipe. After roof collapse, further degradation can be exacerbated by mass movement, so that the hydraulic action of the resulting concentrated flow in the collapsed pipe feature will start at a relatively advanced stage. On

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agricultural fields, the soil loss from such pipe-collapse gullies can be considerable (Poesen *et al*., 1996; Torri *et al*., 2002; Chapter 1.12), generating significant off-site effects (e.g. floods with considerable sediment yields and deposition). Agricultural clearances quickly become unproductive on affected land and, because such landscapes often seem irrecoverable at this point, they are often referred to as 'badlands' (Bryan and Yair, 1982).

2.6.2 TYPES OF PIPING IN EUROPE

There are three distinctive contexts within which pipes can initiate in Europe (Figure 2.6.1): piping in the Gleysols and organic peats (Histosols) common in upland areas in northern Europe; piping in the collapsible loess belt of north and central mainland Europe; and piping in the dispersive sodic marls of southern Europe.

2.6.2.1 Piping in Histosols and Gleysols of Upland Rural Areas in Northern Europe

In a survey of UK piping, Jones *et al*. (1997) found that 70% of piping was located on the Histosols of upland rural areas, mostly peats and organic soils with high winter rainfall acceptance potential (WRAP classes 5 and 2), having a capacity to shrink and crack (Coquet, 1998), and with distinctive subsurface impermeability discontinuities which focus pipe flow (Holden and Burt, 2002; Holden *et al*., 2002). Jones *et al*. (1997) argued that as much as one-third of UK agricultural land could be affected. The mechanisms and dynamics of UK piping in peaty and organic Histosols are covered elsewhere, and are not the primary focus of this review (Jones, 1981; Jones, 1990; Bryan and Jones, 1997).

2.6.2.2 Piping in Collapsible or Destructured Soils of the Northern European Belt[1]

Several authors have described how piping can play a major role in gully erosion on the collapsible or destructured loess soils (Luvisols) of the northern European belt. Collapsible soils are those characterised by a narrow size-range distribution (in the silt size category). The material is vulnerable to erosion because of this low clay content, a situation exacerbated by low organic content. In a dry state, the physical arrangement of loess particles is normally characterised by a low bulk density. However, upon wetting, there is some rearrangement of particles resulting in a loss of volume and corresponding increased density (A. Solé-Benet, CSIC, Spain, personal communication; Derbyshire *et al*., 1995). The cracks that are produced in this change of state act as macropores which focus infiltration, which, because of the very erodible poorly structured nature of the materials, can lead to pipe development along these failure planes. The failure planes will focus throughflow, causing piping especially at gully heads (Collinson, 1996, 2001), and this process may have a role in the backwearing rate of gully heads in those locations. This may be an important explanation for so-called 'bank gullies'.

Factors affecting pipe or tunnel erosion in loessic materials are well researched from work in northern China by Zhu *et al*. (2002), where (as in Europe), failure planes are common in loess over 2 m thick. Botschek *et al*. (2002a,b) found that in shallower loess soils in Nordrhein-Westfalen, a 'duplex' soil profile, characteristic of Luvisols generally, was a necessary prerequisite for the pipes to develop a preferential macropore route. Winzen (2002) and Winzen and Botschek (2003) described and mapped extensive areas of pipe activity on the

[1]Although loess is the most frequently cited collapsible material, Solé-Benet observed that upon dissolution of gypsum minerals, some gypsum soils can also experience compactions and collapse, which induces piping, and so could be placed in this category (A. Solé-Benet, CSIC, Spain, personal communication).

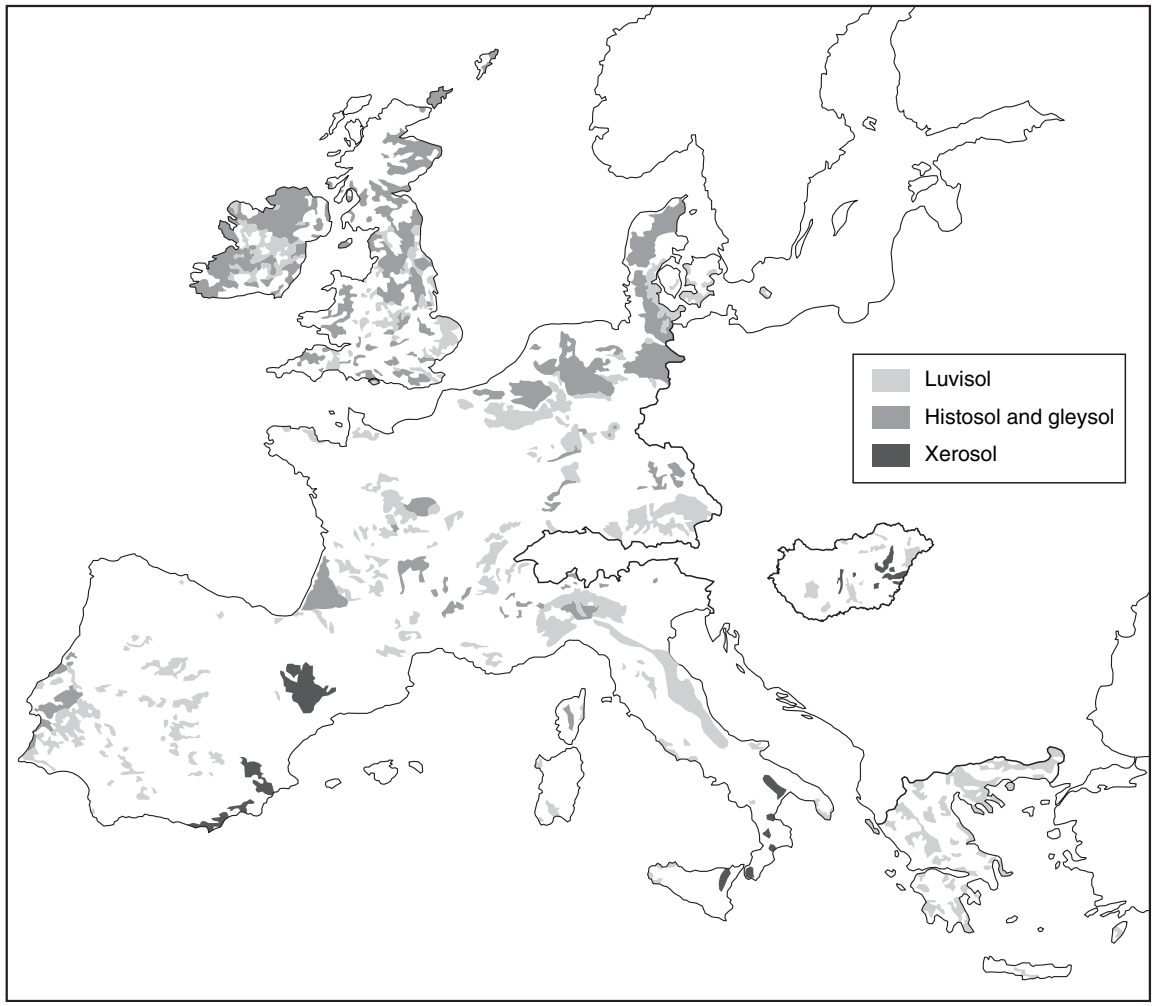
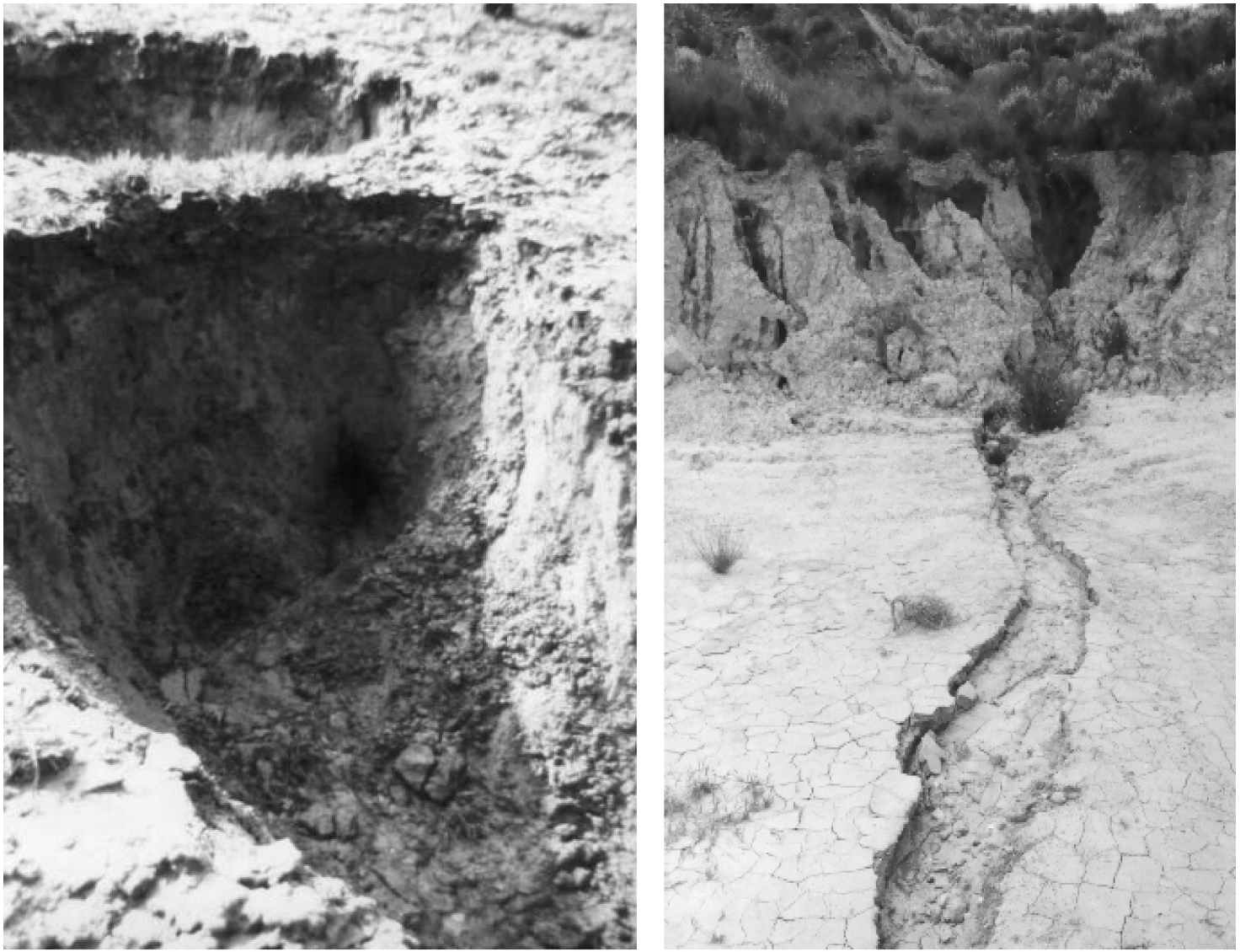


Figure 2.6.1 Distribution of three types of piping-prone materials in Europe *(data as available May 2004):* Luvisols of the north and central European Loess belt; Histosols (peats) and Gleysols of uplands in Northern Europe; dispersive Xerosols of the southern Mediterranean

loessic belt in Germany (www.tunnelerosion.de). In Bergisches Land of Nordrhein-Westfalen alone, over 100 sites of extensive piping activity have been mapped (Hardenbicker, 1998; J. Botschek, Institut für Bodenkunde, Universität Bonn, personal communication; Figure 2.6.2a).[2] In the Ukraine, an area of 50 000 km^2 of collapsible, weakly structured loessic material is seriously at risk (Kevenji, 1994), and on loessic materials in northeastern Hungary, Gábris *et al*. (2003) found that where slope gradients exceeded 12°,

[2]It is important to note that in the investigations of Botschek *et al*. (2002a,b) in loess, no relationship between base saturation, ESP (exchangeable sodium percentage), SAR (sodium absoption ratio) or EC (electrical conductivity) and pipe activity was found, distinguishing pipes on the northern and eastern European loessic belt from those on sodic marls in the south of the Mediterranean basin, where clay dispersion plays a significant role.



(a) (b)

Figure 2.6.2 (a) Sinkhole and bridge in collapsed pipe, northern Germany (Photograph reproduced with kind permission of B. Göbel). (b) Extensive subsurface vertical pipe development in dispersive marls exposed in the side of a man-made quarry, Mocatan badlands, south-east Spain (note otherwise good vegetation cover).

piping and gully erosion occurred, covering over two-thirds of the cultivated area. The role of loess has been recognised in Belgium (Poesen 1989; Poesen *et al*., 1996), particularly in historical gully erosion, and Poesen has estimated that the extent of piping-affected area in Belgium may be over 1000 km^2 (J. Poesen, KU Leuven, Belgium, personal communication).

2.6.2.3 Piping in Dispersive Materials in Southern Europe

The third context for piping in Europe is represented by the sodic and dispersive marine-sourced marl sediment in semi-arid or Mediterranean climates of several central and southern European sedimentary basins (Figure 2.6.1), where the scale of the general involvement of dispersion-origin pipe development in rill initiation and gully erosion has been the focus of considerable recent research (Lopez-Bermudez and Romero-Diaz, 1989; Benito *et al*., 1991; Calvo-Cases *et al*., 1991, 1992; Calvo-Cases and Harvey, 1996; Calzolari *et al*., 1993; Gutierrez *et al*., 1997; Farifteh and Soeters, 1999; Torri *et al*., 2002; Faulkner *et al*., 2000, 2003a,b,). For Spain, Solé-Benet, currently estimates the potential area of materials affected by dispersion to be several thousand square kilometres, concentrated in the Ebro basin and in the south-east of the country, of which perhaps 1000 km^2 are in Almeria (A. Solé-Benet, CSIC, Spain, personal communication). For Italy, on the Pliocene deposits, Torri has estimated that 8000 km^2 are affected (D. Torri, CNR, Italy, personal communication), but this could be tripled if medium-risk deposits are included, hence the total for Italy may be over 20 000 km^2.

In contrast to the processes operating in collapsible soils, dispersion is a physico-chemical process affecting a subset of so-called 'double-layer' clays.[3] All 2:1 clay minerals are constituted by a sandwich of one octahedral layer between two tetrahedral layers, and can be of several types: talc and pyrophyllite, micas, vermiculites, smectites, etc. However, certain double-layer clay minerals (e.g. smectite) are physicochemically 'active' in the presence of saturating levels of monovalent cations (e.g. sodium). The effect of monovalent cations preoccupying exchange sites on the layers of these particular clays is to alter layer charge so much that clay molecules deflocculate, because the bonds that initially existed between clays and other larger particles no longer exist (A. Solé-Benet, CSIC, Spain, personal communication). Several authors have emphasised how the threshold separating flocculation and deflocculation affects both the hydrological and erosional response of the soil (Agassi *et al*., 1981; Sumner, 1992; Levy, 2000; So and Aylmore, 1995; Mualem and Assouline, 1992; Shainberg, 1992). Additionally, swelling on wetting can also be extreme provided that clay minerals are volumetrically significant in the soil or material texture. Hence the term 'activity' in this context refers to physical changes (deflocculation, changes to material permeability due to swelling, loss of soil structure and erodibility) resulting from increased repulsive surface charge in the presence of sodium and especially during wetting. In effect, a chemical process is causing complex physical changes.

The geochemical properties that predispose towards pipe development in dispersive materials have been fully explored by Sumner and Naidu (1997), and this text should be referred to for a full development of dispersive soil theory. For diagnosis of reactive clay states, it has been noted that the soilwater solution of dispersive soils has a high sodium adsorption ratio (SAR) generally >3, and electrical conductivity (EC) below a threshold value for flocculation. Naidu *et al*. (1995) provide a wealth of examples of applications of the well-known threshold plot between EC and SAR that separates dispersive from merely potentially

[3]Or 2:1 clays, where this ratio refers to the number of tetrahedral and octahedral layers forming the basic structural unit of the clay. All clays of the type 1:1 and many of the 2:1 type are considered not active. Moreover, this is mostly due to the particular arrangement of the clay platelets (and not the arrangement of elementary layers) in which the specific mineralogy of the clay type plays a role (A. Solé-Benet, CSIC, Spain, personal communication).

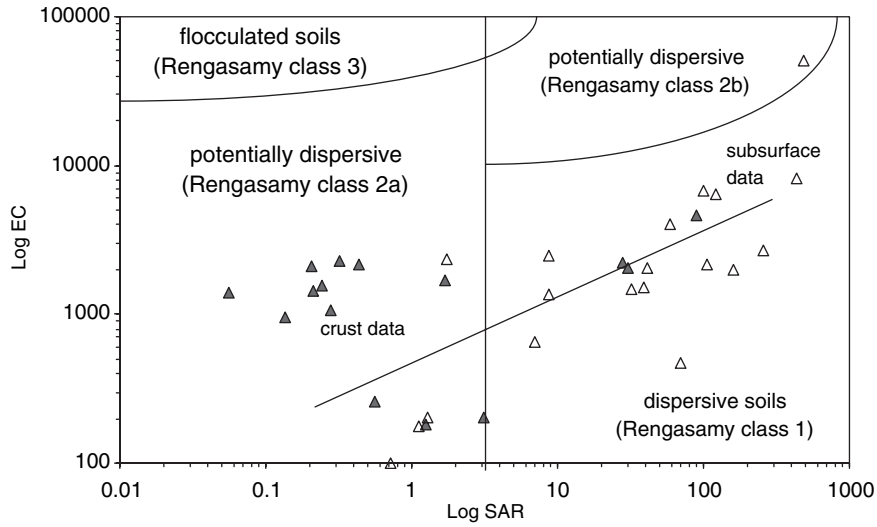


Figure 2.6.3 The log-transformed 'diagnostic' relationship between electrical conductivity (EC) and sodium adsoption ratio (SAR) as plotted for the crust and subsurface samples of local soils in the Mocaton catchment, south-east Spain. Crust samples are filled triangles Note that domains of 'activity' as defined by Rengasamy *et al*. (1984) suggest that the crust is largely non-dispersive, despite a very dispersive subsoil (SAR $\gg$3). (Reprinted from Faulkner *et al*., The role of some site geochemical processes in the development and stabilisation of the three badland sites in Almeria, Southern Spain, *Geomorphology* **35**: 87–89. Copyright 2000, with permission from Elsevier)

dispersive soils[4] (see Rengasamy *et al*., 1984; Rengasamy and Olsson, 1991). Faulkner *et al*. (2000) developed this idea further, demonstrating that diagnostic site 'signatures' (relationships between SAR and EC, particle size and SAR, and pH and SAR) are useful tools in characterising dispersive site morphology and can also be used to distinguish between surface and subsurface materials (SAR/EC function plotted against Rengasamy's classes on log-transformed axes in Figure 2.6.3). Whether or not the gully parent material plots within Rengasamy's class 1 or class 2a,b affects the response of the 2:1 clays in the material to water passing through pores, and thus controls dispersion, pipe development and material availability for erosion.

[4]To measure the effectiveness of sodium in the soil water system, we use the sodium adsorption ratio (SAR):

$$\mathrm{SAR} = [\mathrm{Na}]/([\mathrm{Ca} + \mathrm{Mg}]/2)^{0.5} \tag{1}$$

cations normally in mmol l^{-1}. ESP can be calculated using an ammonium acetate extraction method from soil concentrations, using:

$$\mathrm{ESP} = \frac{(100 \times \text{exchangeable Na})}{\text{cation-exchange capacity (CEC)}} \tag{2}$$

noting that:

$$\mathrm{CEC} = \Sigma(\text{exchangeable Ca} + \mathrm{Mg} + \mathrm{K} + \mathrm{Na} + \mathrm{Al}) \tag{3}$$

here cations normally as mg per 100 g soil (Faulkner *et al*., 2001).

2.6.2.4 Variations with Climate

This variety of predisposing materials produces a fractured pattern of risk-associated scenarios for the intensification of piping from the three types of susceptible materials described so far. Obviously, piping in loess is restricted to the distribution of loess in Europe, and no climatic constraints on this distribution can be inferred. However, examination of the range of settings for pipe activity illustrated by the sample scenarios listed on Table 2.6.1 suggests that piping in Histosols and Gleysols seem to require a *humid temperate* climate. This is because first, high rainfall totals are needed to produce the predispsing Histosols in the first place, and second, an extremely wet scenario is envisaged by Jones *et al*. (1997) for the development of this type of pipe. On the other hand, dispersive-type pipes do not occur in all European depositional marl basins, only those within a *Mediterranean or semi-arid* context. The reason for this appears to lie in the mobility of sodium in a wetter climate. Piping does not develop in exposed marls in northern Europe because sodium is lost so rapidly from the materials by leaching in a humid climate (Churchman and Weissman, 1995) that the dispersive role on the clay complex does not persist. Additionally, in climates with higher rainfall totals, we can assume that the organic matter remains a structuring agent within the topsoil and that organic acids buffer the action of sodium on the clay complex. By contrast, clay is frequently the only structuring agent in drier climates, so its dispersion has a dramatic impact; indeed, Bryan and Jones (1997) observed that pipes in semi-arid areas can be up to four times larger those in other climatic areas.

Imeson and Emmer (1992) explored the impact of climatic change on soil organic matter and on the chemical composition of the soil. For climatic change scenarios envisaged for the southern Mediterranean basin, they argue that a general deterioration in soil structure is to be expected, and that ''... particularly vulnerable are silty soils, soils with a duplex character, and soils susceptible to dispersion''. In this context of change, the piping-driven erosion of dispersive soils is a particular threat on the sensitive Mediterranean/semi-arid desertification threshold. However, until recently the complexity of piping in dispersive materials was not part of desertification models, perhaps because less was understood about clay mineralogy, sodium and the role of dispersion in piping than is understood about non-dispersive piping and gullying. This topic is consequently given particular attention in the remaining sections of this chapter.

2.6.3 PIPE DEVELOPMENT

2.6.3.1 Macropores and Hydraulic Gradients

In the case of collapsible soils, a low clay content and very high silt percentages lead to very low aggregate stability, exacerbated by very low organic carbon content. Erosion thresholds are consequently extremely low in horizons with these characteristics. When pore sizes are sufficiently large, routeways are rapidly eroded into pipes (Figure 2.6.4). Hence, if a destructured loessic landscape has sufficient relative relief for hydraulic gradients to be maintained as erosion proceeds, the enlargement of pipes along preferential infiltrating pathways will be rapid.

In the case of dispersive soils, the development of macropores into preferential routeways and the ease with which the material can be removed in those routeways are facilitated not by the innate structureless nature of the materials (as with loess), but by the mineralogy and hence the behaviour of the material's clay fraction, specifically the double-layer clays (e.g. smectite) in response to sodium on the cation-exchange sites. As was discussed above, sodium, especially high exchangeable sodium percentages (ESPs), will deflocculate double-layer clay particles and thus render the material very erodible. The way in which a double-layer clay can liquefy at critical sodium concentrations was demonstrated 30 years ago by Sherard and Decker (1976) (Figure 2.6.5).

TABLE 2.6.1 A range of European scenarios, representing a climatic transect of sites, within which piping erosion associated with collapsible and dispersive soils might typically dominate (see Figure 2.6.1)

Site	Soil	Lithology	Climate	Land use	Problems	Possible solutions
UK, N-W Europe	Histosol	Peat	Marine temperate	Upland grazing	Gleysols and Podsols Over impermeable substrate leading to pipe development and erosion	Land management - use regulation
Belgium, N-W central Europe	Haplic Luvisol	Loess	Marine temperate	Cultivated land pasture	Collapsible soil	Management of down-slope field borders, banks, etc.
Hungary E central Europe	Aquic Distric Eutrochrept Luvisols on loess	Keuper marls Loess	Temperate humid	Forest, arable, and pasture	Dispersive and collapsible soils, flooding triggered by pipe flow	Management of vegetation and water table
Spain, S-E mainland Med. Europe	Xerosols Luvisols Cambisols	Marls	Semi-arid	Olive groves, Vineyards	Cracking soils affected by piping and erosion	Appropriate land management
Lesbos, Greece S-E Med. Europe island	Lithosols Cambisols	Volcanic lava and pyroclasts	Semi-arid, dry Sub-humid	Grazing on Abandoned terraced vineyards and olive grove	Subsurface water piping, water piping and terrace collapse	Grazing control, vegetation cover, water runoff control
Italy, S-E Med. Europe	Vertisols Entisols	Marine marls Ligurides outcrop	Mediterranean Sub-humid	Arable land, Pasture	Dispersive soils, Badlands, levelled badlands	Bio-remediation, conservation, salt tolerant plant associations
Portugal, S-W Med. Europe	Eutric Leptosols Calcaric Cambisols, Regosols.	Marine marls (mudstones), Plio-quaternary silty sediments	M. semi-arid	Clearance, Irrigation, Terrace Abandonment	Dispersive and collapsible soils, Terrace destruction, Loss of irrigation water, Badlands	Local changes in land management Conservation, Natural vegetation improvement

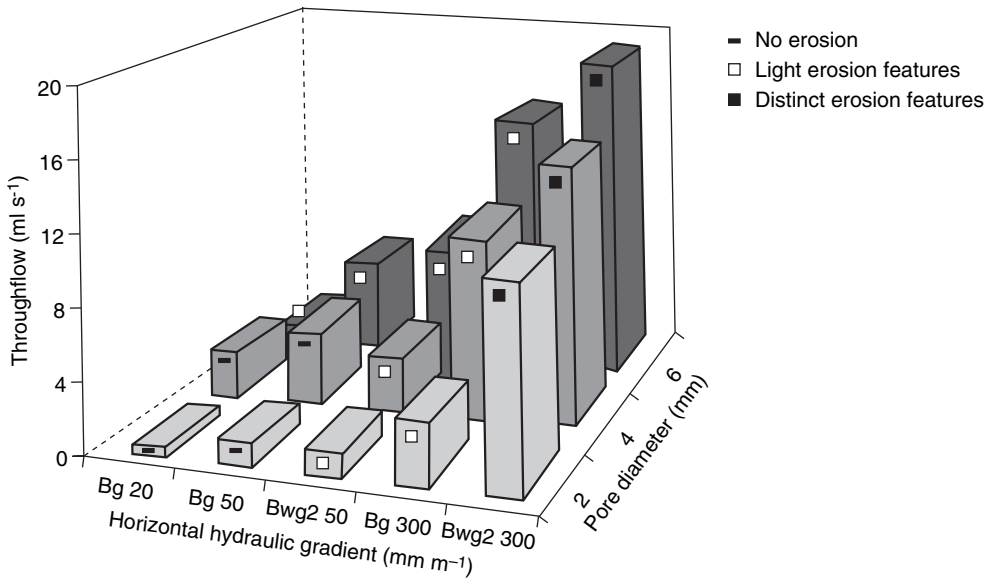


Figure 2.6.4 Pipe erodibility of soil samples from an upper (Bg) and a lower (Bwg2) horizon of a soil profile in Loess. (Botschek *et al*., *Journal of Plant Nutrition and Science*, Vol. 165 (4), Hydrological parameterization of piping in loess-rich soils in the Bergisches Land, Nordrhein-Westfalen, Germany pp. 506–510, 2002, with permission from Wiley-VCH)

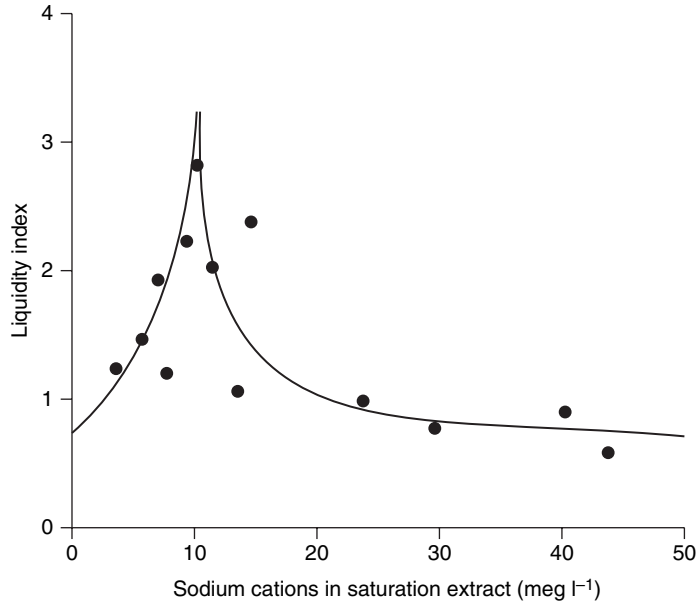


Figure 2.6.5 Liquidity index variation as a function of the pore water's sodium content. (Copyright ASTM International. Reprinted with permission)

Along infiltration surfaces, following hydraulic gradients within increasing connected macropore networks, entrainment and erosion are rapid even at the low stream powers involved.

However, the whole process is complicated by various subsidiary criteria, such as how dominant the clay representation in the material is and the swelling and shrinkage properties of the clay (dependent on the clay representation again, and the clay mineralogy), which together determine the degree to which the swelling of the clays will cause surface cracking on shrinkage. Another consideration is the extent to which clay is preferentially translocated down profile along those gradients, and the role that this plays in rendering the subsurface impermeable and/or in changing horizon variations in clay content, because this will enhance the development of 'duplex' conditions which predispose the concentration of flow into those preferential pathways. Other criteria are the nature of the crustal response to rainfall and the potential that the whole process has for translocating not only clay, but also sodium, down through the material, changing its subsequent response to those same processes through time, as this may eventually create a non-dispersive surface layer (material stabilisation).

2.6.3.2 Shrinking and Swelling in 2:1 Clays and the 'Duplex' Condition

Many badland regolith materials are characterised by shrinking and swelling behaviour (see Rooyani, 1985, for definitions). Imeson (1986) asserted that a soil containing upwards of about 30–44 % of clays with the 2:1 arrangement can both swell and shrink, which can cause surface 'polygonal' cracking and a surface with a 'popcorn' appearance (Figure 2.6.6). Between defined swelling and shrinking limits, the increase in volume is

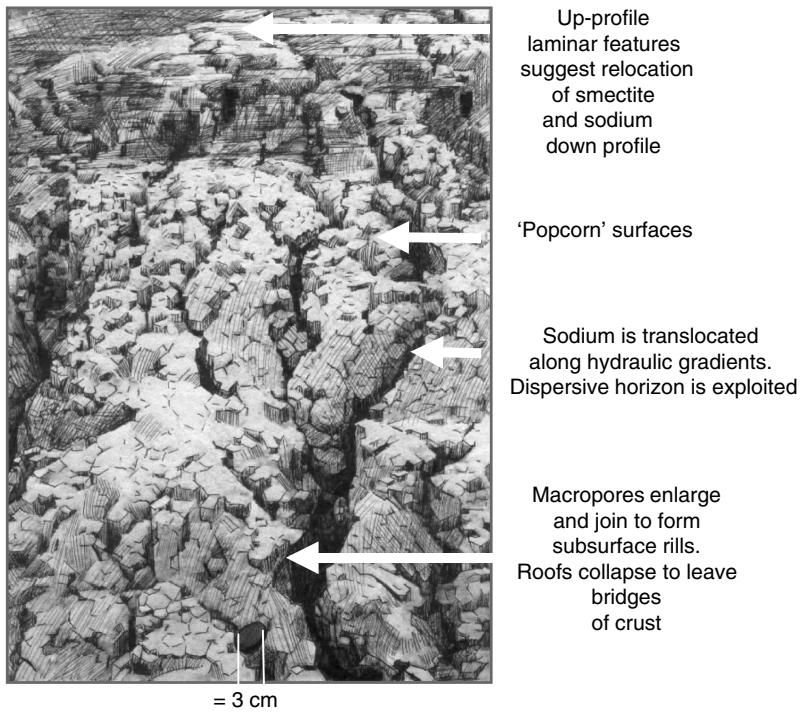

Up-profile laminar features suggest relocation of smectite and sodium down profile
'Popcorn' surfaces
Sodium is translocated along hydraulic gradients. Dispersive horizon is exploited
Macropores enlarge and join to form subsurface rills. Roofs collapse to leave bridges of crust
= 3 cm

Figure 2.6.6 Micro-forms indicative of a dispersive context for rill development. (From Variations in soil dispersivity across a gully head displaying shallow subsurface pipes, Faulkner *et al*., *Earth Surface Processes and Landforms*. Copyright 2004, John Wiley & Sons, Ltd, reproduced with permission)

more or less proportional to the increase in volumetric moisture content. Additionally, there is often hysteresis whereby the amount of swelling during the wetting is less than preceding event shrinkage, leaving cracks open and allowing egress of infiltrating water, which might be assumed to enhance infiltration rates (Faulkner, 1990).

However, in soils with a very high clay content, this effect is less than one might expect, especially if the subsurface is impermeable, because the swelling of the outside of peds or aggregates lowers the hydraulic conductivity so much that draining rainwater can only realistically pass through the system along preferential flow paths in the period before the soil has time to swell – then it becomes blocked by subsurface swelling. Also, the effect of shrinkage cracking is often offset by the effect of surface slaking. In this sense, despite dispersive conditions and desiccation cracking, where clay content is high and hydraulic gradients suppressed, piping will not proceed and the hardsetting of the surface will enhance overland flow and surface wash of the Hortonian kind (Tomlinson and Vaid, 2000).

Lopez-Bermudez and Romero-Diaz (1989) have made the connection between the translocation of clay down profile in dispersive materials and differentiation of clay content with depth in these materials. Figure 2.6.7 (Imeson and Kwaad 1980) demonstrates the typical 'duplex' character of many affected soils, possibly an effect exacerbated by clay relocation down-profile. The soil's duplex nature is important because soil or sediment horizons with slightly differing clay contents will experience *differential* swelling and shrinkage in wet and dry periods. Changes in macropore permeability at these textural discontinuities has been shown by Imeson (1986) to be associated with tensions generated by different swelling rates. In the example in Figure 2.6.8, although the shrinkage ratios (slope of line) and the shrinkage limit (intersection) are relatively similar, differences in the end of each line show that different horizons have different swelling limits. Whatever the cause of the variations in clay content, this differential swelling sets up stresses in response to structural planes of weakness that act as macropores and focus throughflow and pipe enlargement in particular horizons. For these reasons, pipes locate at significant subsurface textural discontinuities in so-called 'duplex' soils. These are commonly referred to in the Australian literature (Rooyani, 1985; Fitzpatrick *et al*., 1995), and the effect has also been noted in southern Africa by Reinks *et al*. (2000).

Some early work (Heede 1971) suggested that the change from vertical to horizontal directions in pipes can be related to water table variability. Unless there is a substantial spring snowmelt (as in some Alpine semi-arid climates), water tables are normally very low in the environments we are discussing, so this effect is unlikely to be very important. It is more common that macropore enlargement is principally connected with small- and medium-scale

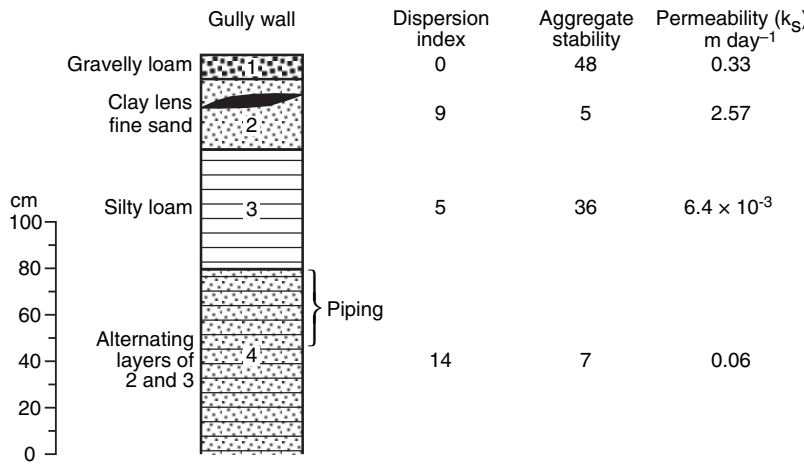


Figure 2.6.7 Sketch of material exposed in gully side near Koubi, northern Morocco. (Reproduced from Imeson AC, Kwaad FJPM, *K.N.A.G. Geografisih Tijdschrift* 1980, **14**: 430–441, with permission from Royal Dutch Geographical Society)

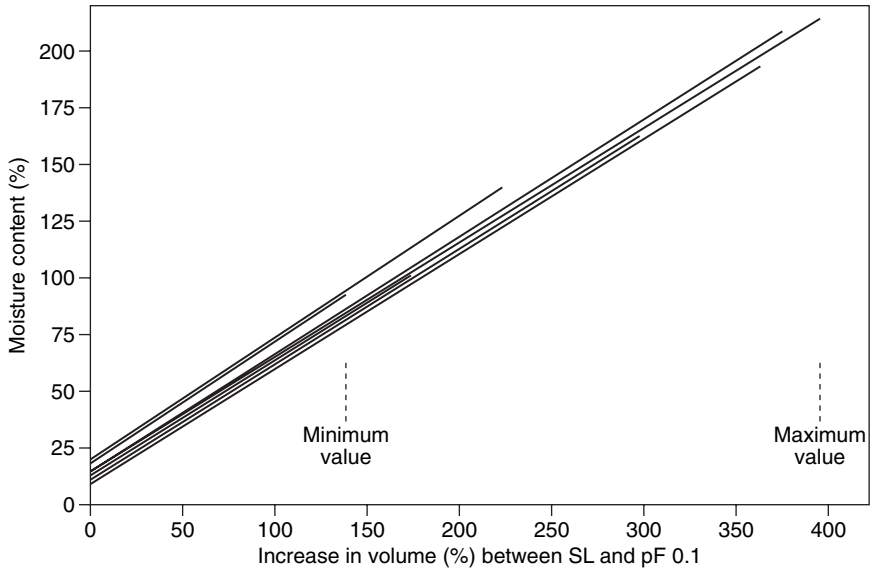


Figure 2.6.8 Line lengths represent differing increases in volume when adjacent badland soils of slightly differing minerealogies are allowed to swell freely. SL = shrinkage limit; pF = capillary potential. (Imeson AC, *Zeitschrift für Geomorphologie*, **60**: 115–130, reproduced by permission of Gebruder Bornträger Verlags. http:/www.schweizerbast.de)

structural features such as those found in duplex soils, although many authors have emphasised the very important role of mass movement failure planes as being responsible in many locations for pipe positions (Campbell, 1989).

2.6.3.3 Dispersion and Infiltration

As we have seen, even at very low rainfall intensities, certain sodic clays with a susceptible mineralogy and a significant clay content can rapidly slake and seal at the *surface*, reducing surface infiltration rates (Sumner and Stewart, 1992). Rengasamy and Olsson (1991) also found dispersion and deflocculation of clays causes swelling during infiltration, reducing the permeability *within the whole soil mass*. However, because swelling only affects the clay content, the material's overall particle size distribution is a critical determinant of these infiltration rate effects, i.e. only soils with a significant percentage of 'active' clay in their composition experience the very adverse effects on infiltration capacity rates demonstrated in Figure 2.6.9. By contrast, in sandy/silty soils, with a relatively low clay content, dispersion of those clays merely renders soils more erodible as a result of the loss of their only structuring agent, in which case infiltration rates are not really affected by swelling, and soils will continue to pipe at depth along macropores (Faulkner *et al*., 2000). At this relatively smaller scale, macropore distribution and the hydraulic gradients produced within the macropores then become the critical determinant of pipe extension. These are the main presdisposing settings for piping, especially where topography allows. To emphasise the point again, a high 2:1 clay content will *restrict* pipe enlargement (Kazmann, 1985; So and Aylmore, 1995), a low or moderate 2:1 clay content but high silt will *enhance* piping in dispersive settings, provided that suitable hydraulic gradients exist through the site.

Unfortunately, the percentage clay content threshold values separating the above two scenarios have not been accurately defined, but are likely to vary with clay mineralogy. However, Faulkner *et al*. (2000) suggested that the 'signature' plot between particle size and SAR can be an important diagnostic tool to be used along with the SAR/EC threshold when assessing the dispersive character of materials.

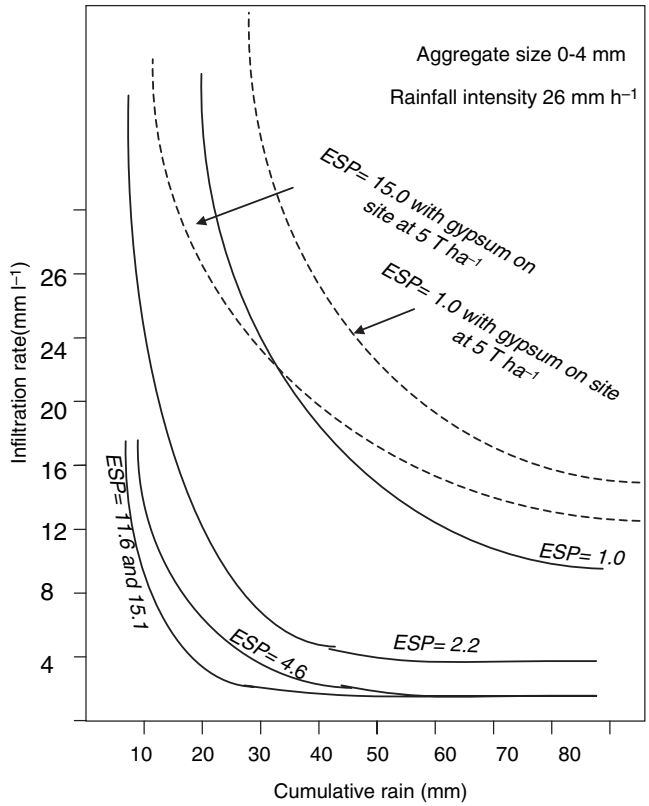


Figure 2.6.9 The negqative effect that the sodium content of the soil (measured here as exchangeable sodium percentage, ESP) has on the infiltration curve shape on a single soil sample. Various amendments with gypsum are shown to have a beneficial effect on soil deflocculation and therefore infiltration. (Reproduced from *Soil Crusting: Chemical and Physical Processes*, Sumner ME, Stewart BA. (Eds.), Chemical and mineralogical components of crusting, Shainberg, I, pp. 33–54, 1992, Copyright Taylor and Francis Group, LLC)

2.6.3.4 Runoff Generation and Sediment Entrainment

Apart from infiltration effects, a whole range of other physical process thresholds are significantly affected by gechemical dispersion properties (Bouma and Imeson, 2000). Imeson and Verstraten (1981) have also explained that the well-documented correlation between erosion and rainfall intensity occurs not only because of rainfall energy, but also because during a rainfall event variations in the soil solution salt dilution occur, complicating (but largely enhancing) deflocculation and dispersion. The significance for sediment entrainment and erosion is clear from Figure 2.6.10, where it can be seen how that the dispersion threshold is enhanced and dispersion persists during a flash flood in Morocco.

2.6.3.5 The Role of Dispersion in Soil Crusting

The surface slaking effect is usually held responsible for the development of '*hard-set*' *crusts* (Bradford and Huang, 1992; Mualem and Assouline, 1992), which are known to restrict surface infiltration (Sumner and Stewart, 1992; Phillips and Robinson, 1998). However, other work suggests that even in soils with a dispersive signature at depth, there can be a loss of sodium at the surface of these materials through time, eventually

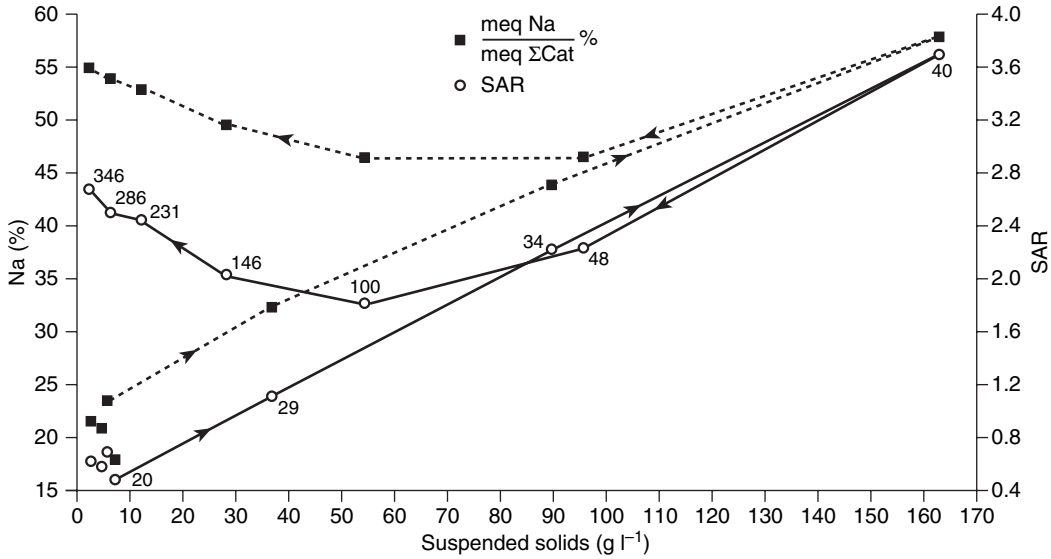


Figure 2.6.10 Relationship between suspended solid concentrations and ESP and SAR values during a flood at Kala Iris, Morocco. The numbers along the continuous line indicate time in minutes since start of runoff. (From Suspended solids concentrations and river water chemistry, Imeson AC, Verstraten JM, *Earth Surface Processes and Landforms*. Copyright 1981, John Wiley & Sons, Ltd, reproduced with permission)

rendering the crust non-dispersive (Faulkner *et al*., 2000, 2001). Torri *et al*. (1994) and Torri and Bryan (1997) likened these *non-dispersive crusts* to a *weathered* '*rind*'. Some crusts may be *polygenetic*, as for instance on the Tortonian marls in Tabernas (Solé-Benet *et al*., 1997).

Algal crusts may stabilise a dispersive material (Alexander and Calvo-Cases, 1990; Alexander, *et al*., 1994). In the presence of a vegetation cover also, surface stabilisation occurs when free hydrogen ions in organic acids allow hydrogen exchange with sodium. When vegetation is not present, surface stabilisation is usually conceptualised as occurring during leaching, when calcium replaces sodium on the exchange complex (Armstrong *et al*., 1998), although there is no doubt that some calcium must also be moved up and redeposited or exchanged within the crust during evaporation. Both processes are probably involved in the development of a *calcic crust,* but the evidence from most sites is that a 'top-down' model explains soil development best. For example, Lopez-Bermudez and Romero-Diaz (1989) record not only a loss of dispersive status in the topsoils on marls in south-eastern Spain, but also a translocation of clays down-profile. This has been noted in a wide range of environments. Imeson and Kwaad (1980) (Figure 2.6.7) for instance, give examples of soils where movement of clay down-profile affects layer permeability in a Moroccon soil, focusing pipe development above the impermeable layer. This may be described as a 'duplex' soil condition.

2.6.4 MORPHOLOGICAL CHARACTER OF PIPE-ORIGIN LANDSCAPES IN EUROPE

2.6.4.1 Small Scale: Shallow Subsurface Pipes, Rills and Bridges in Dispersive Settings

If infiltration is persistently reduced at depth by translocation of clays and enhancement of material swelling, and we have a cracked, calcic, non-dispersive crust, it is possible to argue that infiltration remains fairly good

at the surface but is deflected into a shallow subsoil zone, which may preferentially erode (Faulkner *et al*., 2004). The deeper translocation of clay and of sodium has been observed by several authors (Sherard and Decker, 1976; Imeson and Kwaad, 1980; Romero-Diaz and Lopez-Bermudez, 1989; Alexander *et al*., 1994), and this effect further exacerbates the deflection of throughflows into the shallow subsurface. This was the inferred explanation for shallow pipes in dispersive marls in the Ebro basin, Spain, given by Benito *et al*. (1993) and for shallow pipe development in the biancana badlands in Tuscany by Torri and Bryan (1997). In the Tabernas badlands in Almería, which has a generally non-dispersive upper soil with little evidence of piping, Solé-Benet *et al*. (1997) noted that wetting during infiltration was restricted to a shallow upper soil layer. It was also noted that the crust (of various kinds in Tabernas) was relatively resistant to erosion, and that sediment was preferentially produced from micro-rills in the regolith, rather than from the surface. This would fit the logic of the arguments given by Imeson and Verstraten (1988) for subsurface liquifaction in the shallow subsurface zone in the Rio Frades badlands, Spain.

Faulkner *et al*. (2004) hypothesise that shallow subsurface dispersion is operating even in materials that are only potentially dispersive (SAR$<$3) at depth. In Almerían badlands, layering in the top horizons of dispersive marls can commonly occur by translocation of ions and clay minerals along infiltrating pathways, to produce a non-dispersive crust with a suprisingly dispersive immediate subcrusted layer. This observation is used to explain the appearance of a certain type of crusted rill (Figure 2.6.11) which develops in the shallow subsurface zone, and displays bridges, discontinuity and excessively high depth-to-width ratios. Since both Torri *et al*. (1987) and Gerits (1991) found that a significantly lower transport threshold was needed to move sediment in laboratory experiments on dispersive soils (i.e. in Rengasamy class 1 zone in Figure 2.6.3), the dispersion in this layer would seem to go some way in explaining the morphological characteristics of these rills.

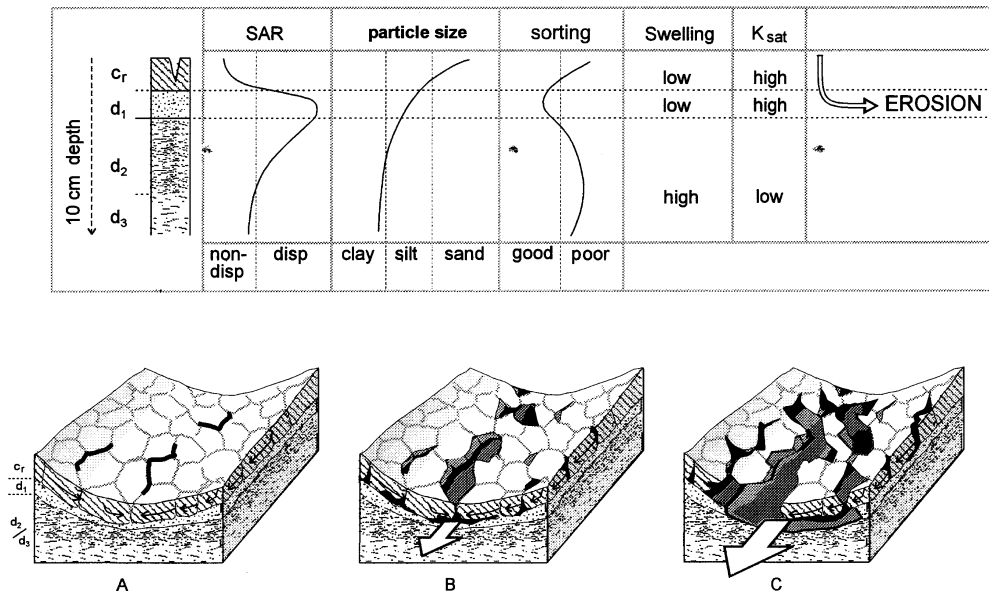


Figure 2.6.11 Summary of relationships found between various soil properties and depth across a small gully head in dispersive marls in south-east Spain. The relationship between these properties and the development of shallow subsurface pipes and rills is suggested. (From Variations in soil dispersivity across a gully head displaying shallow subsurface pipes, Faulkner *et al*., *Earth Surface Processes and Landforms*. Copyright 2004, John Wiley & Sons, Ltd, reproduced with permission)

2.6.4.2 Medium Scale: Gully Networks

Gully development, mass movements and collapse can be significant secondary consequences of pipe enlargement. As we have seen, horizontal layers developed in marls to produce a 'duplex' condition can deflect infiltrating water to particular horizontal pathways. This condition is not typical of collapsible loess, although after prolonged pedogenic evolution an argillic horizon may produce similar 'duplex' character. One way or another, a 'duplex' state can be commonly noted in most pipe-prone materials, including peaty histosols and in loess, and provided that there is a sufficient hydraulic gradient and an outfall site, pipes on these routeways will develop. Pipe erosion is further enhanced by other pre-existing macropores, which become the preferential flow paths. These can be tectonic joints (Benito *et al*., 1993; Farifteh and Soeters, 1999), tension cracks, desiccation cracks, animal burrows (Pickard, 1999) and root channels.

Certain topographic settings promote the growth of macropores into pipes because of the way in which hydraulic gradients are affected. Prerequisites are (a) an infiltrating surface, (b) convergent flow paths to a lower outfall site and (c) convex morphology, this being needed to maintain positive hydraulic gradients within horizontal sections of a pipe net. Hence a considerable relative relief is necessary for the full development of a pipe network in all materials (Botschek *et al*., 2000). In propitious settings, where convergent flow to a suitable (lower) hydraulic outfall is ensured, subsurface erosion enlarges pores, which consequently act as improved drains – a positive feedback mechanism (sometimes referred to as seepage erosion or 'sapping') which ensures that hydraulic gradients will remain high as pipe erosion proceeds. Along these routeways, deflocculated clays will be cumulatively entrained at relative low stream powers, rapidly enlarging macropores into pipes (Fitzpatrick *et al*., 1995). In fact, any natural textural change with depth may deflect pipes horizontally, as has been described for duplex soils (Rooyani, 1985) and particular vertic soils (Swanson *et al*., 1989) in relation to pipe development. Thus, in most cases, pipes will tend to follow the hydraulic gradient (Baillie *et al*., 1986) and flows can emerge at the surface at gully banks, road cuts or (depending on lithological variations and macropore patterns) even at mid-slope locations within the convexity (Heede, 1971; Fitzpatrick *et al*., 1995; Gutierrez *et al*., 1997; Faulkner *et al*., 2003a), and it follows that extensive pipes do not form in low gradient concave hillslopes even if the material is highly dispersive (Ternan *et al*., 1998).

Behind the outfall, pipes can develop into complex branched networks, whose planform morphology mirrors the hydraulic pathways through the site (Terzaghi and Peck, 1966; Farifteh and Soeters, 1999). Poesen *et al.* (1996) demonstrate how on convex sites in agricultural and reworked settings (sites 1, 2 and 4 in Figure 2.6.12) pipe networks can start as discontinuous elements within an otherwise surface network, their pattern frequently influenced by agricultural constructions such as lynchets and tillage. Because levelled land produces a low bulk-density site, any convexity is prone to pipe development (Clarke and Rendell, 2000), and in fact Borselli *et al.* (Chapter 2.12) and Faulkner *et al.* (2003a) have both demonstrated that when materials are collapsible or dispersive, agricultural land levelling and/or terracing terracing can exacerbate rather than reduce erosion. In the *biancane* badlands in Tuscany, Torri *et al.* (2002) and D. Torri (CNR, Italy, personal communication) found that if the top pedogenized horizons (low sodium content) are eroded for any reason, the percolation water that arrives at the dispersive lower horizons has a lower EC, closer to that of rain water. Hence it is more aggressive and pipes can start to develop, a process that is further enhanced by the presence of cracks in the upper soil horizons. In Almeria and in the Ebro basin in Spain, the problem of pipe development and growth on several reworked sites has sometimes become extreme. Especially behind agricultural terraces which provide the predisposing topographic convexity, land may be irrecoverably degraded by pipe erosion (Guttierrez *et al.* 1997). Garcia-Ruiz *et al.* (1997) have explored the additional possibility that irrigation on levelled terraces in these types of materials, especially using water having a low electrolyte concentration, may seriously enhance this type of subsurface erosion.

Parts of pipe systems can rapidly integrate into large nets, sometimes by internal capture. During the full development of a pipe network, pipe roofs can collapse suddenly, revealing the great size of the pipes

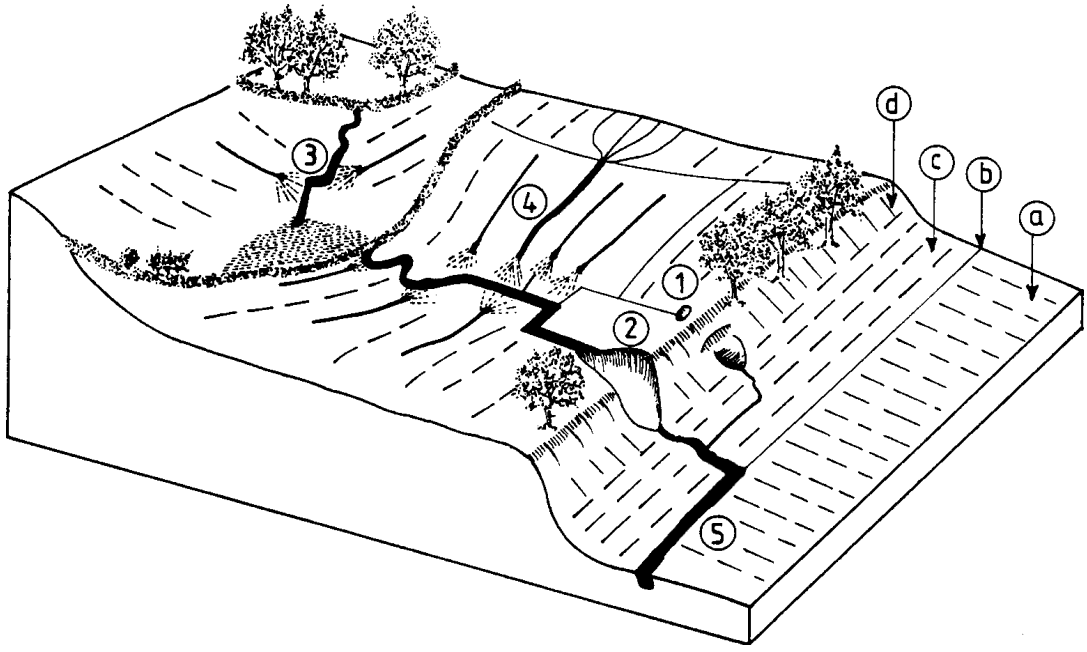
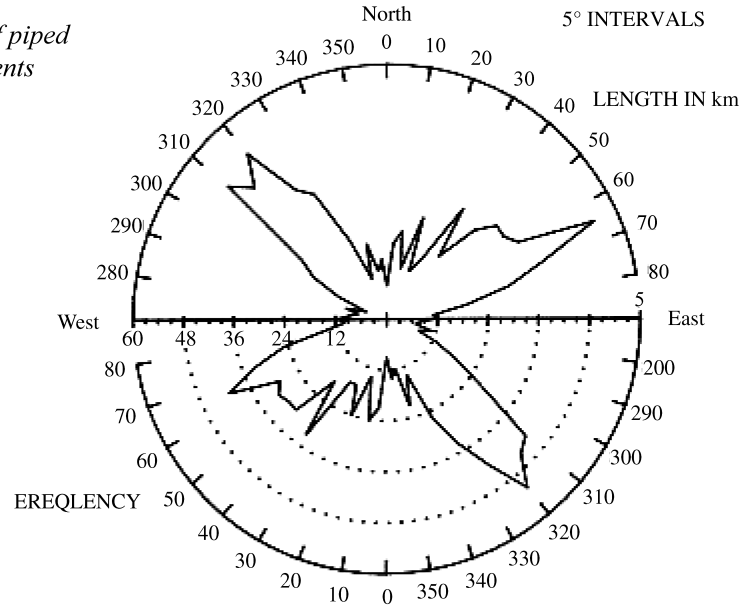


Figure 2.6.12 Sketch illustrating various gully types on agricultural lands in the Loessic belt of Belgium. 1 = Pipe roof collapse feature; 2 = bank gully (possibly collapsed pipe?); 3 = ephemeral gully in valley bottom; 4 = ephemeral gully in valleyside; 5 = ephemeral gully in linear landscape element; a = tillage direction; b = limit of headland; c = headland; d = bank (lynchet). (Reproduced from Poesen *et al*., in *Erosion and Sediment Yield: Global and Regional Perspectives*. Proceedings of the Exeter Symposium, July 1996. IAHS Publication No. 236, 1996, with permission of IAHS Press)

and forming a gully feature into which surface flows now converge (Figure 2.6.2). Often the degraded pipe erodes back by virtue of the stream power of surface flows converging on the degraded pipe-collapse features. Gullies of this size can have a 'hammer-head' form and, depending on hydraulic gradient development, can reach tremendous sizes. Working in the Basilicata area in Tuscany, Italy, both Colica and Guasparri (1990) and Farifteh and Soeters (1999) found that joints in the clayey bedrock are the main source which provide inlets, paths and outlets for infiltration and flow of the water, resulting in the removal of the soil particles underground. In Figure 2.6.13, the morphometric investigation of lineaments in the pipe nework produced by the latter authors displays a very strong structural control on the drainage pattern.

Steep-sided gullies formed by pipe collapse often experience mass movements at their heads (Collison, 2001), since the shear strength of affected materials decreases drastically once macropore enlargement has occurred. These failure planes focus further concentrated flow, and pipes can develop in mass movement failure surfaces very quickly, especially since the bulk density of failed material is so low. A gully in a landslide failure plane in the Mocatan catchment, formed from the collapse of a pipe, has been measured at over 10 m deep, but only 1.7 m wide. Additionally, excavated pipes will evacuate slope materials very rapidly if suddenly coupled at interior nodes to aggressively degrading channels at their base; such coupling events can be envisaged to produce very high sediment yield. A pipe system in the Mocatan basin was observed in only 2 years to widen by 3 m and cut back by 8 m into a gully bank in that time. By comparison, Hortonian processes in extensively piped environments are basically cosmetic.

Rose diagram of piped network lineaments



Piped network lineaments in plan form

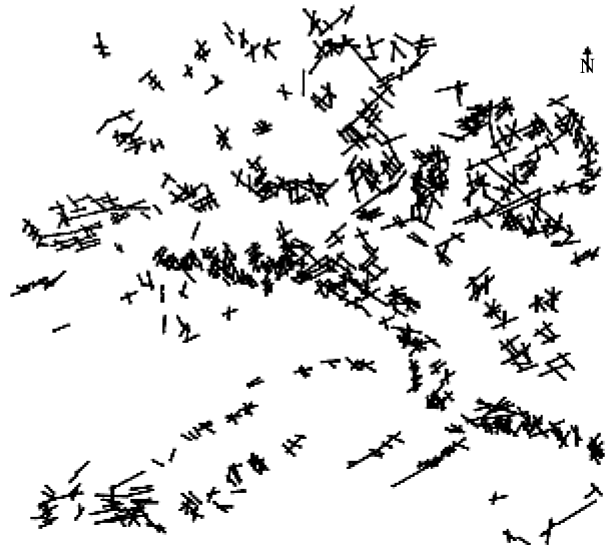


Figure 2.6.13 The relationship between piped network element orientation and two preferential fracture patterns in native materials in the landscape is suggested by the rose diagram. (Reported from *Geomorphology*, Vol 26, Farifteh J, Soeters R, Factors Underlying piping in the Basilicata region, Southern Italy, pp. 239–251. Copyright 1999, with permission from Elsevier)

2.6.4.3 Large Scale: Badlands

2.6.4.3.1 Evolution of Badlands in Dispersive Materials: Variability with Materials, Relative Relief and Connectivity: Almeria Case Study

Bouma and Imeson (2000) recently explored the generalised relationships between such morphological indicators and erosion processes in the Petrer badlands, Spain. With increasing general agreement emerging, it is possible to argue that, with a good knowledge of local structure and with a good topographic map base, the pattern of development of a pipe network could be predicted from a given set of initial conditions. However, at present, no mathematical model of pipe development in dispersive materials has been published. Whether pipes will continue to develop or infill and flare the slope back in time depends principally on the pattern of site hydraulic gradients, themselves dependent on connectivity. Connectivity itself is dependent on the rejuvenation history age of the whole landscape over a range of time-scales from very recent to Holocene or longer (Harvey, 1982).

Most badlands appear to be in marls or mudrocks that are dispersive (some would dispute this), but certainly not all display active piping. Why not? Does piping diminish in intensity with time? And how generalised is the role of wash in badland evolution? Process measurements are few in badland areas, and are almost prohibitively difficult to undertake in piped landscapes, yet without process data this question is difficult to address. The above discussion suggests strongly that the excavation of a dispersive site by pipe network extension following rejuvenation, for example, will set in train rapid degradation of a landscape. Logic suggests that such very young badlands will have very high yield, episodically focused around changes in connectivity, and characterised by very rapid morphological changes in which mass movements and piping are dominant. An example of this type of badland is found in the Mocatan area of Almeria described previously. However, there are suggestions from Campbell (1989) and Harvey (A. Harvey, University of Liverpool, personal communication) that piping and mass movements are only a landscape erosion stage, and that in the absence of subsequent changes to slope–channel coupling, there may be a later point where surface dispersion is lost (as has been described by Robinson and Phillips, 2001), which would allow the slopes to become increasingly wash dominated. This interpretation is suggested for the older Vera badlands in Almeria (Faulkner *et al*., 2003), and certainly although fresh material at Tabernas has high ESPs and SARs (Alexander *et al*., 1994), the site is so old and has experienced so little connectivity change in the last 10 000 years that the whole landscape has been stabilised, both geomorphologically (no change to hydraulic gradients) and in terms of surface material stabilisation. The landscape is now wash-dominant, although rates of sediment production are very low (Cantón *et al*., 2001), a fact belied by the 'erosive' look of that terrain.

It is tempting to infer that the three badland types represent an *ergodic* sequence, but actually Faulkner *et al*. (2001) found that this is an over-simplification, because the clay content of these sites differs too, denying such a simple model for the Almeria sites. Figure 2.6.14 suggest that any event (intensity and duration) affecting these sites has an effect that varies with material and surface properties. If clay rich, surface sealing will ensure a wash-dominant morphology. If infiltration persists, if the material is dispersive but the clay fraction not too dominant, extensive piping can proceed, especially if relief is high. Otherwise, moderate relief gives rise to a landscape with only limited potential for piping even though the material is dispersive (as at Vera). If clay percentages are less suitable for pipe development, we obtain the wash-dominant – or shallow pipe scenario – once again. Hence diagnosis of material and a lithological and topographic survey to establish connectivity and drainage history of a site are the two vital first stages in interpretation of the morphology, behaviour and future management possibilites of dispersive badland areas.

2.6.4.3.2 Badlands and Environmental Management

As suggested in a review of badland research in the context of climatic change by Torri *et al*. (2000), the term 'badlands' has been recently extended to include areas where piping, mass-wasting and fluvial processes

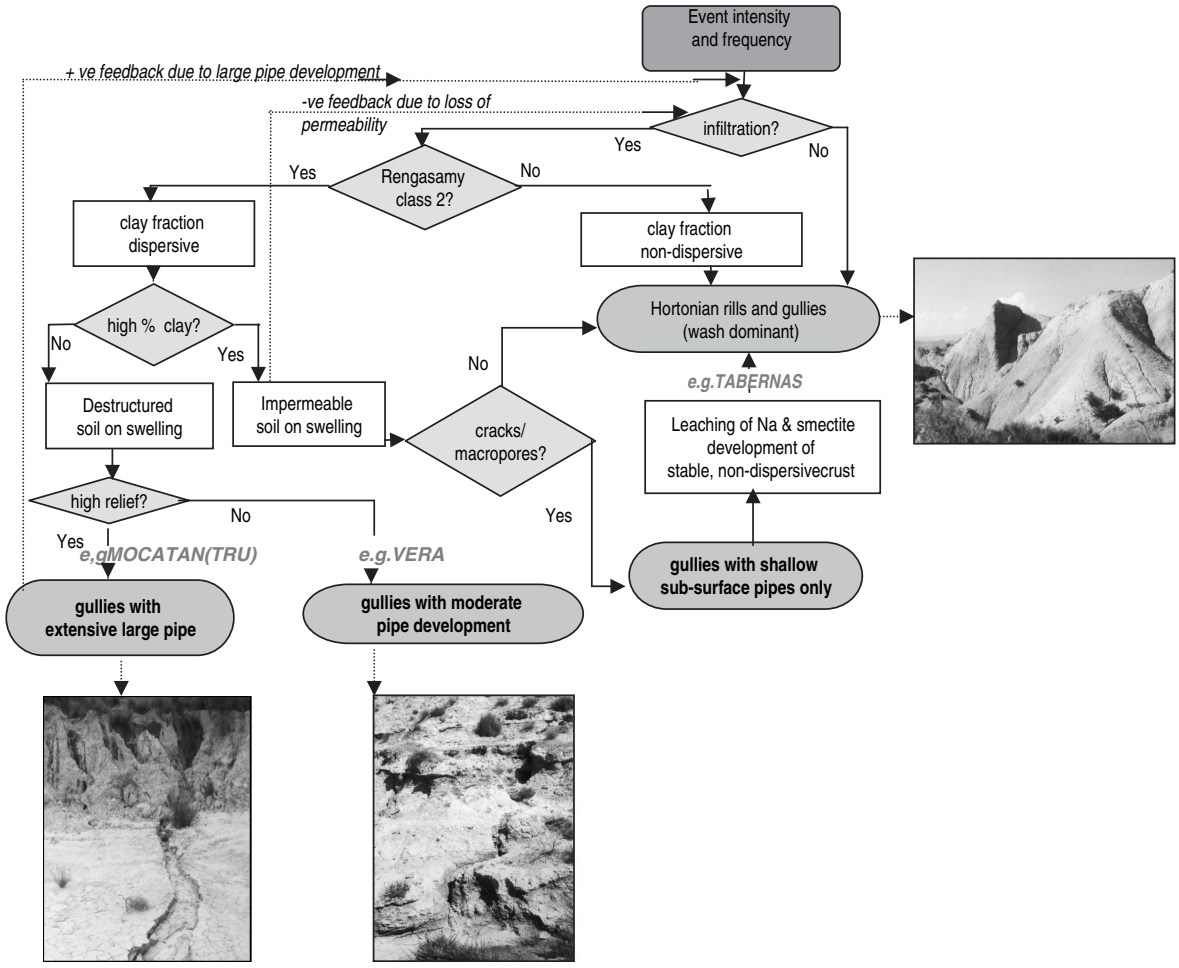


Figure 2.6.14 An explanation of the morphological differences found on the three badland areas in south-east Almeria in terms of dispersive material properties (SAR and clay content of parent material) and relative relief. The photographs illustrate the general appearance of slopes in Mocatan badlands, the Vera badlands and the Tabernas badlands

interact producing '. . . a rugged, hummocky and dissected topography', and area in which land use is '. . . [in] competition between soil erosion, soil forming processes and vegetation'. They go on to suggest that '. . . Badlands are densely dissected areas, which have been severely degraded and where soil has disappeared or lost most of its fertility. The combined effect of climate and badland use prevents the soil from forming or recovering its fertility and erosion prevails'.

Although aggressive badlands seem irrecoverably degraded from the agricultural point of view, in some settings land-use change decisions may be able to change the overall progression of the evolution of badlands from piped systems. Calzolari *et al*. (1997) and later Torri *et al*. 2000 argue that, geologically speaking, the badlands in Italy may fluctuate in intensity through time (hundreds to thousands of years) because the Italian climate is on the semi-arid/Mediterranean threshold which could be seen to be in balance between favouring and depressing vegetation (Phillips and Robinson, 1998), consequently depressing or enhancing erosion. In such a context, land-use changes can impel the system in one or another direction. For example, borderline

dispersive badlands with moderate slopes reclaimed by dynamite land levelling in the early 1900s (and more recently by bulldozers) did not evolve back to badlands because 40-cm deep tillage has been found to impede the growing of pipes, which are thus periodically destroyed. In such a setting, and although this chapter has already argued that land levelling can enhance pipe development, land management may be sustainable (Torri *et al*., 2000). However, But only in low gradients settings: where slope is such that mass movements are common, then tillage doesn't change the situation, and '... dispersion, piping and mass movements are entangled into a single condition' (Torri *et al*., 2002; D. Torri, CNR, Italy, personal communication). In other words, in less steep terrain and along climatic thresholds, anthropic activities may be the 'butterfly's wing' that separates irreversible soil loss from a sustainable agriculture.

It may be that in Italy, as at Tabernas, in their later stage of evolution, badland systems eventually become self-preserving systems with high resilience. However, as we have seen, changes in coupling can alter all that. Similarly, what about climatic change? The present situation in Italy, for instance, indicates a decreased erosion activity in the *calanchi* area of Atri (eastern Apennine margin), an observation Torri finds consistent with most of the behaviour of badlands throughout Italy (even the *biancana* fields tend to be more vegetated than in the past). The reader is referred to the 2000 *Catena* volume for a much wider debate on this issue, but it is interesting to consider how difficult it will be to develop any simple prognosis for the wide range of materials and terrain types represented by 'badlands'.

2.6.5 CONCLUSION

The discussion in preceding sections has attempted to describe how complex the convergent predisposing contexts for piping erosion can be, given the EU's wide range of climatic, geological, topographic and cultural settings. Although there is less clear information on which to base estimates for all European member states, if Gleysols, Histosols and collapsible and dispersive soils in Europe are taken together, the areal extent of the land seriously at risk in the EU from piping erosion has been estimated to exceed 260 000 km^2.

Emphasis has been placed on the view that all soil erosion by water involves both mechanical and chemical mechanisms. When any portion of the parent material is dispersive, a range of well-known physical processes can increase or decrease by several orders of magnitude when chemical thresholds are crossed. Hydraulic conductivity, infiltration rates, shear strength, overland flow and sediment concentration, rill development and gully growth mechanisms all change, and the changes frequently involve pipe development. However, because dispersion and swelling only affect the clay fraction, clay mineralogy and the percentage of clay in the material play a fundamental role in the suppression or enhancement of piping. Changes in material properties with structure and lithology are therefore critical determinants of pipe and gully positions and size at the small scale. However, because both structural variations and overall morphology both strongly influence 3D hydraulic gradients and flow convergence through the site, they are a fundamental determinant of pipe and gully growth at the medium scale also.

We have also seen how gully nets in both collapsible and in dispersive settings can grow by internal connectivity shifts, in a manner essentially controlled by changes in hydraulic gradients as drainage network connectivity evolves. Although resculpting may seem a suitable as a possible geomorphological approach to land management in this terrain, land levelling of dispersive materials into terraces can produce more problems than it solves (Faulkner, 2003; Chapter 2.12). It is important to conclude that both material diagnosis and topographic mapping are important prerequisites, not only to geomorphological explanation, but also for 'getting gully management right', especially in the context of the sensitive threshold that separates Mediterranean climates from the desertification threatened by a change to a semi-arid climate in southern Europe.

ACKNOWLEDGEMENTS

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2.7

Wind Erosion

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2.7.1 INTRODUCTION

Wind erosion occurs in many arid, semi-arid and agriculturally used areas around the world and is influenced by geological and climatic factors in addition to human activities. Wind erosion leads to land degradation in agricultural areas and has a negative impact on air quality. Dust emission by wind erosion is the largest source of aerosols, which directly or indirectly influences the atmospheric radiation balance and hence global climatic variations (Shao, 2000). Wind carries more fine sediment than any other geological agent. It has been estimated that windblown dust from soil erosion contributes about 500×10^6 t of particulates to the atmosphere each year (Greeley and Iversen, 1985). In view of this fact, it can be concluded that dust is an active factor in the climate system. Model calculations indicate that about 50 % ($\pm$20 %) of the total atmospheric dust originates from disturbed soils, i.e. soils affected by cultivation, deforestation or erosion (Tegen *et al*., 1996).

Wind erosion has been overlooked in the past in Europe as a land degradation process. However, it has received more attention as a process responsible for the creeping decrease in soil fertility and as a source of atmospheric pollution (Oldemann *et al*., 1990; Gobin *et al*., 2003; Warren and Bärring, 2003). This is mainly attributed to the removal of fine particles and organic material, which are the most fertile parts of the soil carrying the nutrients and other agents such as pesticides or herbicides. In addition to creeping degradation, single wind erosion events may result in soil losses of more than 100 t/ha^{-1} and cause considerable on- and off-site damages (Funk, 1995; Goossens, 2003). The main problem with wind erosion in Europe is its perception. Heavy sand storms attract attention by disturbing the public once in a while but, in general, the

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TABLE 2.7.1 Relationships between quantity of wind erosion, effects of erosion and annual soil loss. (Reproduced from Chepil WS, *Soil Sci. Soc. Am.* 1960, **24**: 143–145, with permission of the Soil Science Society of America)

Extent of erosion	Description of erosion	Annual soil loss ($t\,ha^{-1}$)
None to insignificant	No distinct visible effects of soil movement	<40
Slight	Soil movement not sufficient to kill winter wheat in boot stage	40–125
Moderate	Removal and associated accumulations to about 2.5 cm depth, sufficient to kill wheat in boot stage	125–375
High	About 2.5–5 cm removal and associated accumulations	375–750
Very high	5–7.5 cm removal with small dune formations	750–1125
Exceedingly high	More than 7.5 cm removal with appreciable piling into drifts or dunes	>1125

processes mostly happen unnoticed. Chepil (1960) pointed out that annual average soil losses up to $40\,t/ha^{-1}$ are possible without any visible sign of erosion (Table 2.7.1). Erosion and deposition processes both take place on large areas and are therefore difficult to identify. In contrast to water erosion, where the eroded material follows determined paths, wind-eroded material is widely dispersed over the landscape. Furthermore, the direction of transport is subject to changes and in some cases completely the opposite, and so are the areas of erosion and deposition.

Wind erosion is widespread in Europe, from Iceland in the north-west to the Russian plains in the far east. There are diverse sets of geographical extensions, which are affected by wind erosion. The affected areas are lowlands or exposed mountains, the climate is dry or humid, the soils are sand, loess or peat and they are fertile or wastelands. The continental dry conditions in eastern Europe favor wind erosion on large areas. In northern Europe, wind erosion is severe on light, sandy soils of the Pleistocene glacial outwash. Close to the coastlines, wind erosion is principally caused by high wind speeds despite humid climatic conditions. All these regions have in common that inappropriate farming practices have intensified the problem (Wilson and Cooke, 1980; Warren and Bärring, 2003).

The spatial extent of wind erosion has increased in recent decades, mainly caused by changes in agricultural practices. The first reason is the spectrum of growing crops, which has changed to greater proportions of arable land crops. Other factors include disturbances of the soil surface by ploughing and a multitude of tillage operations. The time of highest mechanical stress by tillage operations coincides with the time of highest climatic erosivity in spring. Some more factors influencing wind erosion have been identified:

- The higher level of mechanisation has led to larger fields and in consequence to the removal of hedges and other landscape structures.
- Drainage of arable land has caused faster drying of the soil surface, resulting in decomposition of organic matter and decreasing soil aggregate stability.
- Overgrazing is a significant causative factor in the semi-arid and arid regions, where no other type of land use is possible (Frielinghaus, 1990; van Lynden, 1995; Riksen *et al*., 2003).

The effects of wind erosion are soil deterioration, crop damage and pollution of adjacent areas. Soil deterioration includes the loss of fine material and organic matter, the degradation of soil structure and the loss or redistribution of fertilisers and herbicides. The loss of topsoil is the predominant impact of wind erosion in Europe (van Lynden, 1995). In the worst case, the productivity has declined so substantially that arable land has been removed from production, as reported in Sweden and Poland (Jönsson, 1992; Veen *et al*., 1997). The

TABLE 2.7.2 Some on-site and off-site effects of wind erosion. (Reproduced from Goosens D, On-site and off-site effects of wind erosion. In *Wind Erosion on Agricultural Land in Europe*, Warren A (ed.). EUR 20370. European Commission, Brussels, 2003; 29–38, with permission of D. Goossens)

On-site effects	Off-site effects
Soil degradation	**Short-term effects**
1. Fine material may be removed by sorting, leaving a coarse lag	1. Reduced visibility, affecting traffic safety
2. Evacuation of organic matter	2. Deposition of sediment on roads, in ditches, hedges, etc.
3. Evacuation of soil nutrients	3. Deposition of dust in houses, on cars, washing, etc.
4. Degrading water economy in the topsoil	4. Penetration of dust in machinery
5. Degrading soil structure	5. Deposition of dust on agricultural and industrial crops, ruining their quality
6. Stimulated acidification of the topsoil	
Abrasion damage	**Long-term effects**
1. Direct abrasion of crop tissue, resulting in lower yields and lower quality	1. Penetration of dust and its constituents in the lungs, causing lung diseases and other respiratory problems
2. Infection of crops due to the penetration of pathogens	2. Absorption of airborne particulates by plants and animals, leading to a general poisening of the food chain
3. Stimulated dust emission due to sandblasting of the surface top layer	3. Deposition of heavy metals and other eroded chemical substances to the ground, infecting the soil
Other damage	4. Contamination of surface and ground water via deposition of airborne particles
1. Infection, with pathogens or soil constituents, of adjacent uncontaminated fields and crops	5. Increased eutrophication of surface and ground water
2. Accumulation of low-quality wind-blown deposits on the fields	6. Infection of remote uncontaminated areas, transforming these into new potential sources
3. Building of sand accumulations at the field borders, covering of drainage ditches	
4. Burial of plants	
5. Loss of seeds and seedlings	

most severe damage is reported in the former Soviet Union, where wind erosion removes an estimated 1.5×10^6 ha of cropland from cultivation each year and a much larger area is damaged to some extent (Schroeder and Kort, 1989).

Crop damage is caused by the abrasion of seedlings, the excavation of the roots or the burying of the young plants. The consequences often include additional costs for re-sowing. An overview about on- and off-site effects of wind erosion is given in Table 2.7.2 (Goossens, 2003).

An additional factor is the emission of soil particles due to agricultural practice (tillage emission). This kind of emission takes place even under non-critical conditions (calm or low wind speed) and affects especially the finest soil particles. The entire area of a field can emit this fine material several times depending on tillage frequency. Tillage emission depends on the management practices (type and speed of operation), the soil type and the soil moisture conditions. Research related to this kind of emission is scarce at the moment and there is little knowledge about how agronomic practices affect the generation and composition of dust throughout the year. Studies performed in the USA, Sweden, Germany and the UK were focused on human respiratory problems caused by fine particulate matter and measured dust concentrations only at tractor driver level and in the air near the agricultural operation (Darke, 1976; Batel, 1979; Noren, 1985; Clausnitzer and Singer, 1996).

Data on the extent of wind erosion in Europe are very limited and, if available, are the result of different methodologies of assessment on the national scale, which are not comparable. Consequently, there is no

TABLE 2.7.3 Extent of wind erosion in some European countries

Country[a]	Total (1000 ha)	Light	Medium	Severe
Bulgaria[1]	13			13
Czech Republic[2]	963	397	475	91
Denmark[3]	1000			
Estonia[4]	77			
France[5]	500			
Germany	4154			
Lower Saxony[6]	2000			
Mecklenburg[7]	1004	67	543	394
Brandenburg[8]	1150	533	327	290
Hungary[9]	1400			
Latvia[10]	230			
The Netherlands[11]	97			
Poland[12]	8843	5447	3077	318
Russia[13]	7138	2014	4599	525
Slovakia[14]	154			
Sweden[15]	35			
UK[11]	260			
Ukraine[16]	2200			

[a]References: [1]Ivanov, 1997; [2]Janecek *et al*., 2000; [3]Gross and Bärring, 2003; [4]Reintam *et al*., 2001; [5]Montanarella, 2002; [6]Thiermann *et al*., 2000; [7]Funk *et al*., 1996; [8]Funk *et al*., 2001; [9]Fenyö, 1997; [10]FAO, 1997; [11]Riksen and de Graaff., 2001; [12]CNT, 2000; [13]Larionov *et al*., 1997; [14]SSCRI, 2003; [15]Bärring *et al*., 2003, [16]Dolgilevich, 1997.

uniform map of the occurrence of wind erosion in Europe. Nevertheless, there are two regions where European-wide studies show good agreement: severe wind erosion in the north Caucasus and moderate wind erosion along the glacial outwash from the UK to Poland. The wind erosion risk in the Mediterranean region is regarded as high and over a large extent in an assessment by the USDA (USDA, 2003), whereas a European study showed no risk in this area (EEA, 1998). According to the Global Assessment of Soil Degradation, 42×10^6 ha or 4 % of the European territory are affected by wind erosion (Gobin *et al*., 2003). Unfortunately, most European-wide studies exclude the Russian Federation, where very large areas suffer from severe wind erosion. Soils in the north Caucasus region, for example, have lost about 20–60 % of their upper horizons within 15 years (Larionov *et al*., 1997). Studies on a regional scale produce much higher percentages of the affected area, but the comparability of these data is limited by the differences in methodology and definitions. A summary of the available data on the extent of wind erosion in Europe based on national assessments is given in Table 2.7.3.

Regional wind erosion assessments without a spatial reference to the territory of countries have also been made in Austria (Klik *et al*., 2000), Serbia (Letic and Savic, 2002), Lithuania (Racinskas, 1997) and Spain (Gomes *et al*., 2003). Attendant problems of wind erosion have been appraised in Finland (aeolian reactivation of dune fields) (Kotilainen, 2002) and Portugal (coastal wind erosion).

2.7.2 PROCESSES OF WIND EROSION

Wind erosion occurs when three conditions coincide: high wind velocity, a susceptible surface with loose particles, which can be picked up, and insufficient surface protection by crops or plant residues. The two main categories determining the extent of wind erosion are the *erosivity* of the climate and the *erodibility* of the soil.

Both are influenced by the interactions of various other components, resulting in a high temporal variability of the actual erodibility of a particular site (Lyles, 1988; Warren and Bärring, 2003).

2.7.2.1 Erosivity

The *erosivity* of the climate depends on the wind velocity, the amount and distribution of precipitation and evaporation. Their interactions determine the intensity, frequency and duration of wind erosion events on susceptible surfaces.

The wind regime in Europe is determined by three factors: the large temperature difference between the Polar air in the north and the Subtropical air in the south; the land–sea distribution with the Atlantic Ocean to the west, Asia to the east and the Mediterranean Sea and Africa to the south; and the main orographic barriers – the Alps, Pyrenees and Scandinavian mountain chain. Most parts of Europe are influenced by eastward-moving weather systems and so western surface winds dominate there. The average wind velocity in Europe is highest in the north-west (Ireland, Scotland) and decreases in a south-eastern direction. Areas of the highest wind velocities in Eastern Europe are located between the Urals and the coasts of the Caspian, Azow and Black Sea. Local wind systems are prevalent around the Mediterranean Sea. They are characterised by a constancy of strength and direction over long periods, such as the mistral or the scirocco (Troen and Petersen, 1989).

The wind velocity has a daily course caused by thermal effects, with a maximum in the early afternoon and a minimum at night. The yearly variation of wind velocity shows a maximum during the winter in north-western Europe but, owing to a positive climatic water balance, the moist soil surfaces can resist the wind forces in most instances at that time. The climatically highest erosivity is reached in spring, when high wind velocities appear and the evaporation increases owing to rising temperatures. In summer, the average wind velocity is at its minimum in northern Europe whereas in southern Europe local wind systems gain influence (Figure 2.7.1). Precipitation is highest in northern Europe, decreasing in a south to south-eastern

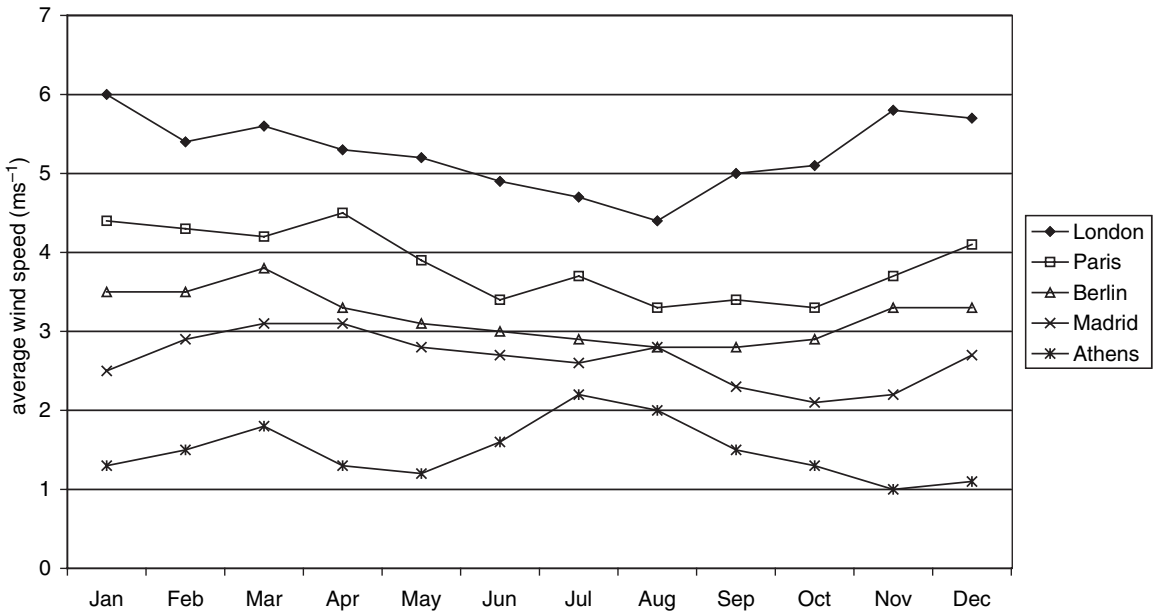

Figure 2.7.1 *Average monthly wind speed for some European cities.* (Reproduced from *World-wide Agroclimatic Database*. FAOCLIM 2. FAO Agrometeorology Group, with permission of the FAO)

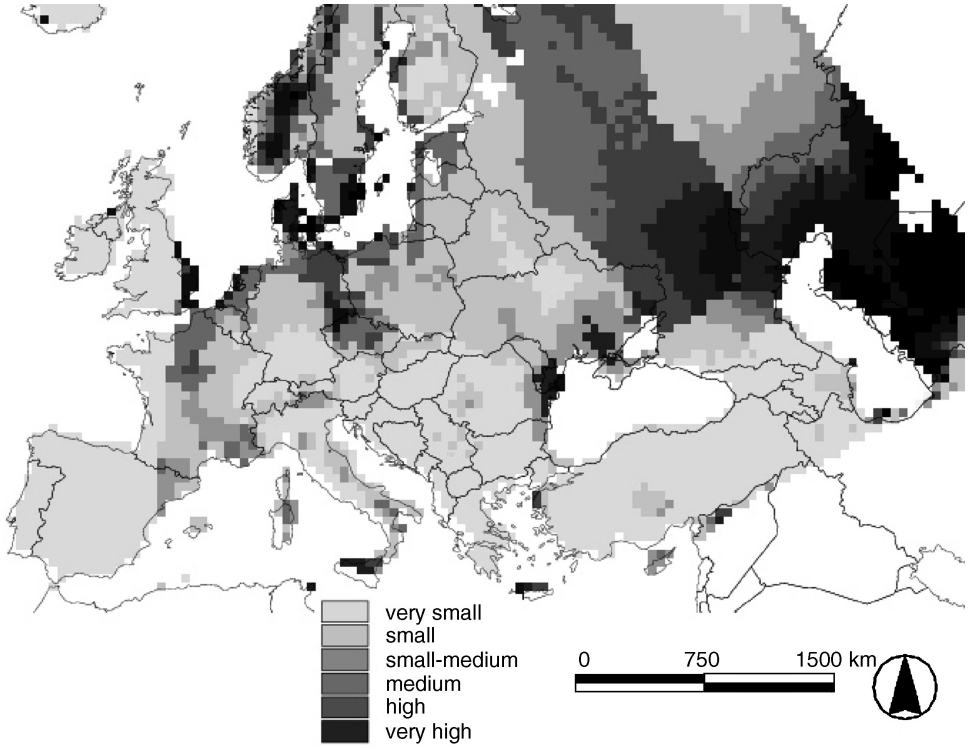

Figure 2.7.2 Wind erosivity of the climate in Europe based on CRU data

direction. The driest regions in Europe are in the south of the Iberian Peninsula and between the Black Sea and the Caspian Sea. The distribution of evaporation is just the reverse of the precipitation pattern, so that two favourable factors for wind erosion coincide in large parts of southern Europe.

An erosivity index for Europe considering the climate factors is shown in Figure 2.7.2. It is based on 30-year average global climate data of the Climate Research Unit on a $0.5 \times 0.5^{\circ}$ grid. This is assumed to represent the 20th century space–time climate variability (New *et al.*, 1999, 2000). The erosivity is mainly determined by the wind velocity and modified by the moisture conditions. This is considered in the erosivity index by the cube of the annual averages of wind velocity ($m\,s^{-1}$] and the ratio of potential evaporation to precipitation.

2.7.2.2 Erodibility

The *erodibility* describes the potential of a soil to erode or, the reverse, the ability to resist the acting wind forces. This is mainly attributed to the texture and organic matter content, which influence the water-holding capacity and the ability of the soil to produce aggregates or crusts (Chepil, 1955). In general, sandy soils are highly erodible because they dry quickly, form only few, weak aggregates and contain a large proportion of particles in the most erodible fraction between 80 and 200 µm. Loamy soils are more resistant against wind erosion but have a greater potential for dust production if they erode.

Natural soils consist of a large variety of grain sizes, which also differ in shape and density. In most cases soil particles are aggregated, with aggregate sizes ranging from decimetres to micrometres. The average

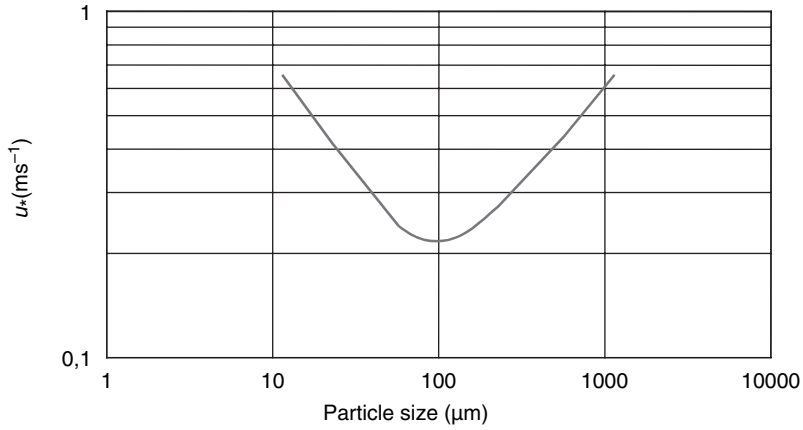

Figure 2.7.3 Dependence of critical friction velocity on particle size (after Bagnold, 1941)

diameter of single grains or aggregates that can be regarded as nonerodible is >840 µm (Chepil, 1942; Woodruff and Siddoway, 1965). These grains are usually too heavy to become airborne by the wind stress. The threshold of motion for uniform grain sizes, expressed in terms of friction velocity of the wind, is shown in Figure 2.7.3. The lowest friction velocity is needed to move particles of size between 80 and 100 µm. The threshold friction velocity increases to the greater diameters caused by weight and to the smaller diameters caused by cohesive forces. The size distribution and stability of aggregates vary within the year and are mainly influenced by soil management practices, hence the erodibility can vary in time and space.

Regarding textural composition, soils with the highest erodibility are located on the sandy glacial outwashes in the northern European lowlands and also on the loess soils of the large aeolian deposits in eastern Europe. The FAO Digital Soil Map of the world has been used to derive an erodibility index by combining soil texture with the organic carbon content of the soils (Figure 2.7.4).

2.7.3 FACTORS INFLUENCING WIND EROSION IN DETAIL

2.7.3.1 Wind

Wind is moving air and is caused by pressure differences in the atmosphere, which in turn result from temperature differences at the Earth's surface. Wind consists of a steady mean part and a superimposed turbulent part. The transport of moisture, heat, momentum and pollutants in the atmospheric boundary layer is dominated in the horizontal direction by the mean wind and in the vertical direction by turbulence. Mean wind is responsible for rapid horizontal transport and can exceed velocities of 100 km/h^{-1}. Turbulence is generated by frictional drag on the air moving over rough surfaces. It results in an irregular swirling motion with turbulent eddies moving up and down. Owing to the increase in wind velocity with height, the net effect of the turbulent motion is always a downward flux of momentum and an upward flux of constituents. Thus, detached soil particles are passed to higher layers of the atmosphere. The magnitude of the vertical wind is about one-tenth of the horizontal velocity (Stull, 1988).

Wind is the driving force of wind erosion if it exceeds a given threshold wind or friction velocity. The latter is better suited to express the momentum flux that the wind exerts on the soil surface and is influenced by the wind and also by surface roughness, as indicated in Equation (2.7.1). Wind velocity increases with height,

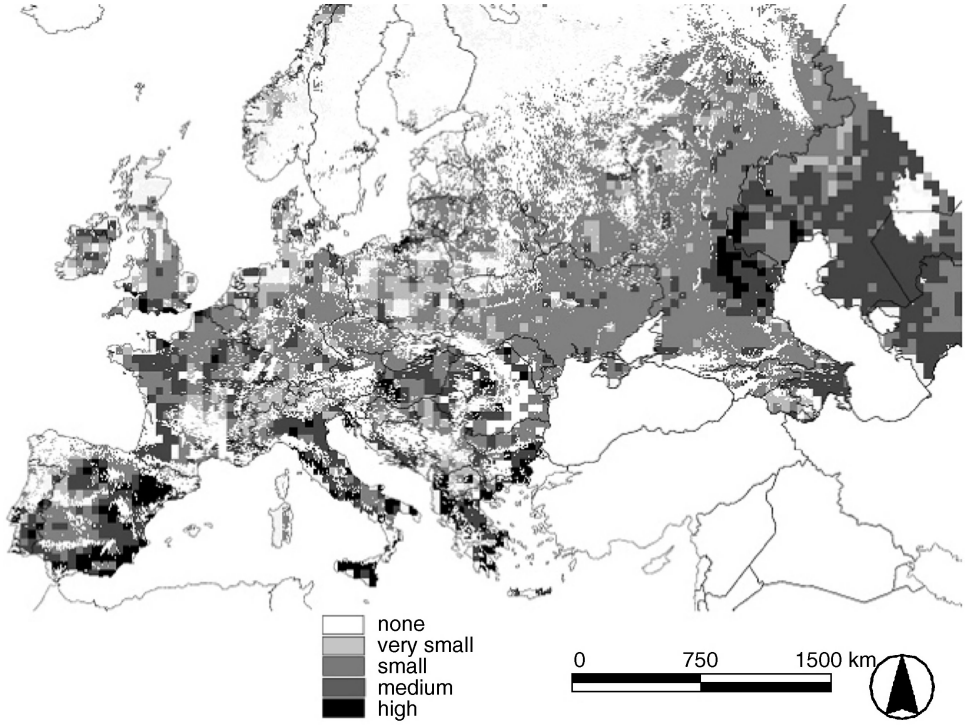

Figure 2.7.4 Soil erodibility in Europe, derived from texture classes and organic matter content, covered with the forest mask (Batjes, 1996; FAO, 2000b, 2003)

whereas the friction velocity is constant within the boundary layer (also called the 'constant flux layer'). The correlation can be described with the logarithmic wind profile as

$$\frac{u_z}{u_*} = \frac{1}{\kappa} \ln\left(\frac{z}{z_0}\right) \tag{2.7.1}$$

where

u_z = velocity at height $z\,(\mathrm{m\,s^{-1}})$

u_* = friction velocity $(\mathrm{m\,s^{-1}})$

κ = Karman constant for turbulent flow (0.4)

z_0 = roughness length, height at which the velocity is zero (m).

A certain wind or friction velocity has to be exceeded to initiate particle movement. This value is referred as the threshold velocity. For loose, unconsolidated grains there is a close relationship between particle size and this threshold value. Bagnold (1941) derived an expression by equating the acting and the holding forces on the uppermost grains of a surface:

$$u_{*\mathrm{t}} = A\sqrt{\frac{(\sigma - \rho)}{\rho} g d} \tag{2.7.2}$$

where

u_{*t} = threshold friction velocity ($m\,s^{-1}$)
σ = grain density ($kg\,m^{-3}$)
ρ = air density ($kg\,m^{-3}$)
g = gravitational acceleration ($m\,s^{-2}$)
d = grain diameter (m)
A = empirical coefficient which depends on the friction Reynolds number, $u_* d/v$, where v is the kinematic viscosity of the air.

After exceeding the threshold value, the transport capacity of the wind increases rapidly and follows an exponential function with exponents between 2 and 4. One of the basic transport equations that describes the relationship between transport rate and friction velocity was also given by Bagnold (1941):

$$q = C\rho u_*^3 \frac{\sqrt{\frac{d_p}{d_{250}}}}{g} \tag{2.7.3}$$

where

q = particle flux (mass per unit width per unit time)
C = a particle size distribution function with values ranging from 1.5 (homogeneous distribution) to 2.8 (heterogeneous distribution)
d_p = particle diameter ($d_{250} = 250\,\mu m$).

As this equation was developed from investigations of dune sands, its application to soils can provide only a rough estimate. The difficulty in establishing an appropriate equation to fit all possible cases is illustrated by the large number of derived equations that are available in the literature. A compilation of mass transport equations was given by Greeley and Iversen (1985).

A comparative assessment of the local erosivity by wind was given by Beinhauer and Kruse (1994), where the daily wind forces were estimated by summarising the hourly wind integrals for chosen threshold wind velocities:

$$SFU_{6,7,8} = \sum_{n=1}^{24} (FF_n - FFT) FF_n{}^2 \tag{2.7.4}$$

where

$SFU_{6,7,8}$ = daily wind force by assumed threshold velocities of 6, 7 and $8\,m\,s^{-1}$
FFT = threshold wind velocity
FF_n = hourly average wind velocity.

Further, the precondition is a dry soil surface, which has at least a 2.5-mm thick top layer. The advantage of this method is the representative comparability of the erosivity of the climate with a minimum of input parameters.

2.7.3.2 Other Climatic Factors

The other climatic factors influencing wind erosion are temperature, humidity, radiation, precipitation and evaporation. They cause temporal changes of the actual erodibility by affecting the soil water balance. In general, wet surfaces are stable enough to resist the wind forces, but for the initiation of wind erosion a very thin dry surface layer is sufficient. The water content of this layer is mainly dependent on climatic factors because the evaporation exceeds the hydraulic conductivity of sandy soils to a significant extent. An estimation of the surface water content can be derived by the comparison of the water content of the top layer (<5 mm) and the evaporation. Inclusion of the surface moisture to the transport equation results in (Skidmore 1986)

$$q = C\left(\frac{\rho}{g}\right)\left[{u_*}^2 - {u_{*\mathrm{t}}}^2 - 0.5\left(\frac{SW}{WP}\right)^2\right]^{\frac{3}{2}} \tag{2.7.5}$$

where

q = erosion rate $(\mathrm{kg\,m^{-1}\,s^{-1}})$
C = parameter (2.5)
u_* = friction velocity $(\mathrm{m\,s^{-1}})$
$u_{*\mathrm{t}}$ = threshold friction velocity of the dry soil $(\mathrm{m\,s^{-1}})$
SW = actual water content of the top soil layer (10 mm) $(\mathrm{kg\,kg^{-1}})$
WP = water content at wilting point (1.5 MPa).

2.7.3.3 Roughness

The term *roughness* is used to describe properties of surfaces ranging from the micro to the macro scale, which represents the effects of single grains and of the topography. The roughness of a soil surface affects wind erosion in two ways. First, a rough surface increases the turbulence and therefore the dissipation of the kinetic energy of the wind at the surface, resulting in a slowdown of the wind velocity (Stull, 1988), and second, the leeward side of clods or furrows is sheltered against wind action and particle impact (Potter and Zobeck, 1988). In addition, the moving material can be trapped and the avalanching increase of transport is impeded.

According to Römkens and Wang (1986), the roughness of a field can be classified in four scales:

1. roughness caused by single grains or aggregates, <2 mm
2. roughness caused by aggregates or clods, <100 mm
3. oriented roughness caused by tillage operation, 100–300 mm
4. topography.

Roughness can be determined directly in the field by measuring height differences with pin meters or laser relief meters (Zobeck and Onstad, 1987; Huang and Bradford, 1990) and indirectly by comparing the foreshortened length of a chain lying on a rough surface with the full length of this chain (Saleh, 1993). Parameters to describe roughness are often the variance, the standard deviation (SD) or the root of the SD of the logarithm of the measured height values (Kuipers, 1957; Allmaras *et al*., 1966). Geostatistical methods are also usable, such as spectral analysis or semivariograms (Linden and van Doren, 1986).

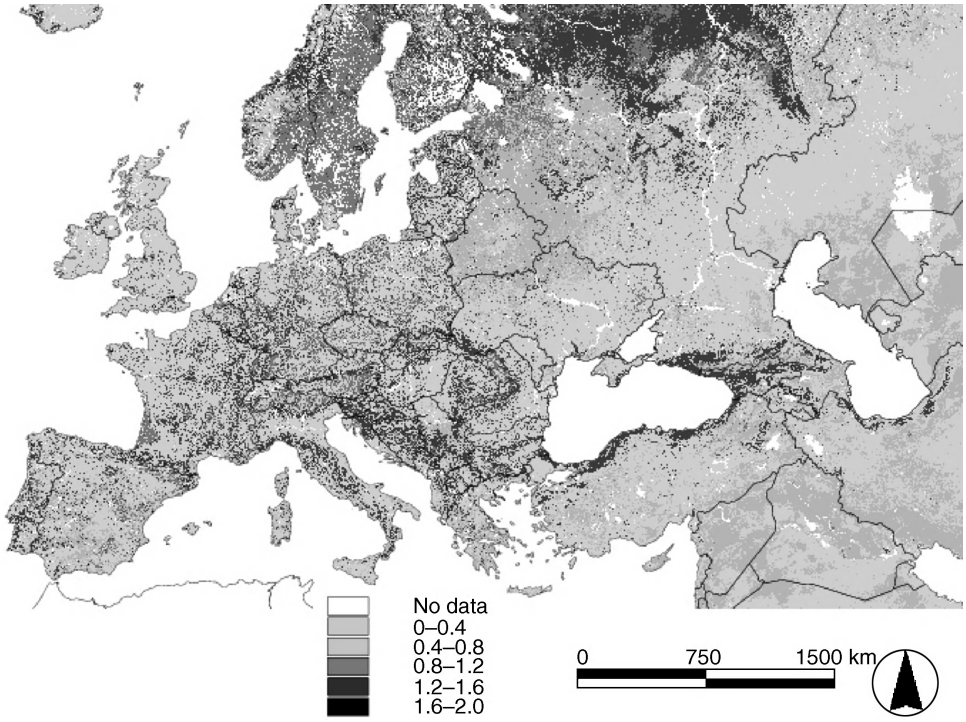

Figure 2.7.5 Aerodynamic roughness length ($z_0$in m) in Europe, derived from CORINE and land cover classification using parameters TA–LUFT (2001)

The roughness of a surface influences the wind profile [see Equation (2.7.1)], so conversely the wind profile can be used to derive information about the surface roughness. The parameter is the aerodynamic roughness length, z_0, which is a feature of the surface and can only be estimated by the wind profile. The wind velocity is considered to be zero at this height. In general, z_0 can be estimated as 0.03 times the average roughness height (Abtew *et al*., 1989). Aerodynamic roughness length classes for most terrain types have been derived by meteorology and can be found in meteorology handbooks.

On the large scale, the roughness of a landscape is determined by the size and distribution of the roughness elements that it contains. These are principally vegetation and built-up areas, which are dominant at the landscape scale. Commonly, the roughness length z_0 can be used to parameterise the roughness of a terrain. Roughness length values for Europe are shown in Figure 2.7.5. The highest values are generally in the mountainous parts. Western Europe is characterized by a heterogeneous distribution caused by a diversified landscape compared with eastern Europe, with more homogeneity and generally smaller roughness length values.

Another concept to determine roughness is the 'shelter angle distribution' (Potter *et al*., 1990), which is referred to the sheltering effect of roughness against particles in saltation and their impact angles.

2.7.3.4 Vegetation

2.7.3.4.1 In the Field

Permanent vegetation is the best measure of protecting soils from wind erosion and plant residues also serve this purpose. The effect of plants on wind erosion can be expressed by the percentage of the surface covered

with non-erodible plant material or by morphometric parameters such as silhouette area or the leaf area index (LAI). The soil cover, in general, refers to flat residues lying on the surface. An equation which includes a wide variety of materials covering the soil surface was given by Fryrear (1985) and expresses the soil loss ratio (soil loss from covered soil relative to soil loss from bare soil) in terms of the soil cover:

$$SLR = 1.81\mathrm{e}^{-0.072\%\mathrm{SC}} \tag{2.7.6}$$

where

SLR = soil loss ratio
$\%SC$ = percentage soil cover.

Equation (2.7.6) is limited to soil cover between 8 and 80 %.

On arable land, the growing crops change every year and therefore there are temporal variations in the protective effect of vegetation according to the time of the year. Measurements have shown that soil covers <10 % increase wind erosion (Morgan and Finney, 1987; Funk, 1995; Sterk, 2000). Soil cover >10 % reduces wind erosion rapidly. The soil loss is reduced to about 50 % at a soil cover of 20 % compared with a bare surface and is prevented completely at soil cover >40 %.

The effect of vegetation on the surface wind can be explained with the principles of fluid mechanics, as roughness elements absorb part of or the entire shear forces from the airflow. As long as the vegetation is sparse and the wind can reach the soil surface, the total shear stress can be split into the stress on the vegetation, τ_v, and the stress on the underlying surface, τ_s (Marshall 1971):

$$\tau = p{u_*}^2 = \tau_v + \tau_s \tag{2.7.7}$$

Within a plant canopy, the drag resistance is proportional to the dynamic pressure $0.5\rho_A u^2$ (where ρ_A = air density and u = wind velocity) and the frontal (silhouette) area, L_A. The shape is taken into consideration by a drag coefficient, C_v. It can be calculated with Equation (2.7.8) by measuring the drag forces, W_v:

$$W_v/F = W_v/NA = C_v\rho_A u^2 L_A/2A \tag{2.7.8}$$

The plant density is given by $A = F/N$, where A is the uniform ground area per plant, F the total floor area and N the number of plants. For row crops, a special case exists (regular arrays) with $A = aw$, where a is the distance between the rows and w the mean distance of the plants within a row. Drag coefficients for plants have been evaluated by Morgan and Finney (1987) and Funk and Frielinghaus (1998). There is another possibility for calculating the drag coefficient from measurements of the wind profile (Morgan, 1989):

$$C_v = \frac{2u_*}{\int_{z_1}^{z_2} u^2 L_A(z)\mathrm{d}z} \tag{2.7.9}$$

Raupach (1992) developed equations to predict the stress partition on rough surfaces for practical applications. The equations were derived for solid roughness elements but they are also applicable to

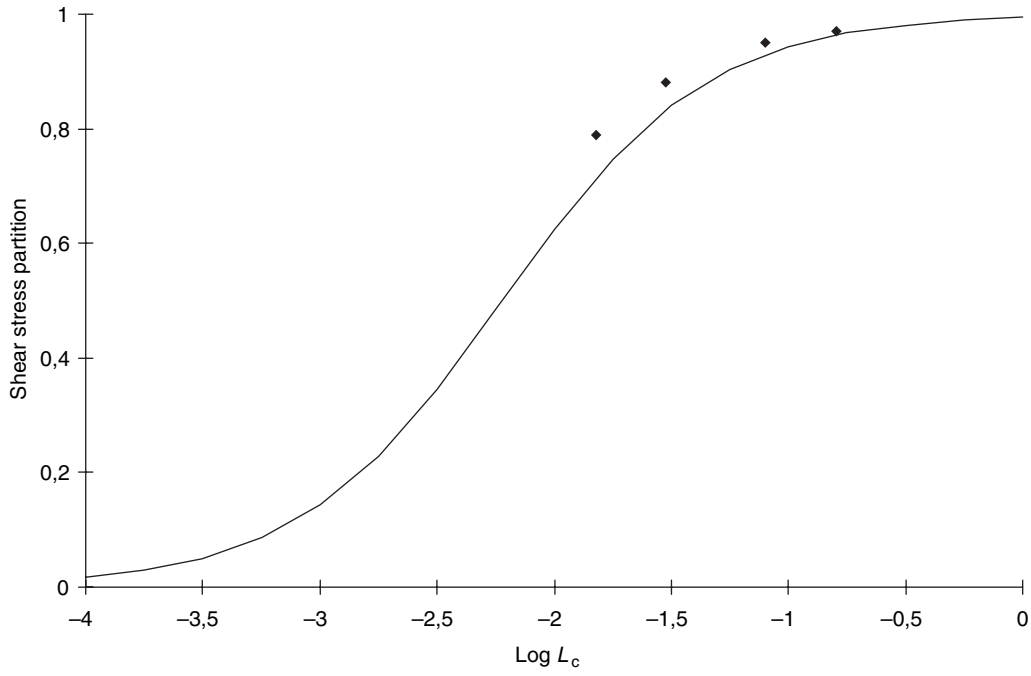

Figure 2.7.6 Dependence of shear stress partition of vegetation (symbols) and solid roughness elements (line) on the lateral cover (plant silhouette area per unit total floor area)

vegetation if its special features are considered. Increasing wind velocity causes changes of the drag coefficients and silhouette areas (Funk and Frielinghaus, 1998). The stress partition given by Raupach (1992) (see Figure 2.7.6) is

$$\frac{\tau_v}{\tau} = \frac{\beta L_c}{1 + \beta L_c} \text{ and } \frac{\tau_s}{\tau} = \frac{1}{1 + \beta L_c} \tag{2.7.10}$$

where

τ, τ_v, τ_s = shear stress (total, vegetation, surface)
β = ratio of drag coefficients C_v/C_s (vegetation/surface)
L_c = lateral cover.

2.7.3.4.2 Around the Field

Shelterbelts are another kind of vegetation used to prevent wind erosion. Especially in the northern European regions with high wind speeds there are many traditional systems of hedges, which protect the fields. Hedges have influences on the local wind field and on many other components of the micro- and macroclimate (Figure 2.7.7). They should best be arranged perpendicularly to the prevailing wind direction. Shelterbelts give protection downwind for a distance of about 25 times of their height, depending on the porosity, kind of trees and number of rows (Nägeli, 1943). Most effective are triple rows, with a tree

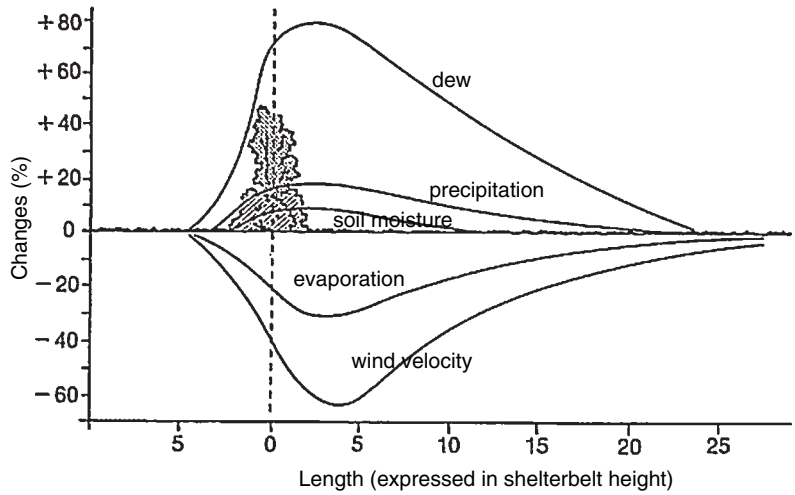

Figure 2.7.7 Effects of shelterbelt height on wind velocity, evaporation, soil moisture precipitation and dew

row in the centre flanked by shrubs, with a triangular cross-section (Chepil and Woodruff, 1963). Increasing permeability with height prevents 'wall effects' on the leeward side. The distances between shelterbelts depend on the erodibility of the soil. Highly erodible soils need a dense network of hedges, which is contrary to an effective work rate of field machinery (Riksen *et al*., 2003). The installation of shelterbelts is fairly expensive, needs a long period of support and becomes effective only after a number of years. Therefore, shelterbelts can be only a supporting measure to prevent wind erosion in combination with measures in the field.

2.7.4 MODES OF PARTICLE MOTION

The modes of particle motion are closely related to the particle size. The density and the shape of particles are also important, if the consideration is extended to agriculturally used soils, which comprise all textural classes and contain organic material with a much lower density than mineral particles of the same size. The first distinction of particle motion was given by Bagnold (1941), who distinguished creep, saltation and suspension (Figure 2.7.8).

Creep. Mineral particles larger than 500 μm are too heavy to be lifted from the surface by wind. They roll or are pushed along the surface by wind or the impact of saltating particles.

Saltation. The bouncing motion of particles across the surface is called saltation. Inactive particles are thrown steeply aloft, accelerate in the higher layers and return to the surface with a small impact angle. The typical lift-off angle is around 55° and typical impact angle is around 10°. The grain size is between approximately 70–500 μm. It is the principal mechanism of transport for large quantities of soil particles in the direction of the wind and amounts to 50–80 % of the total transport. Saltation causes an avalanching increase in transport, if there is sufficient erodible material, until saturation of the transport capacity of the wind is reached. Below saturation, the surplus energy of the bouncing particles works to abrade crusts or aggregates. The height of the saltation layer is often below 1 m (Lyles, 1988; Shao, 2000).

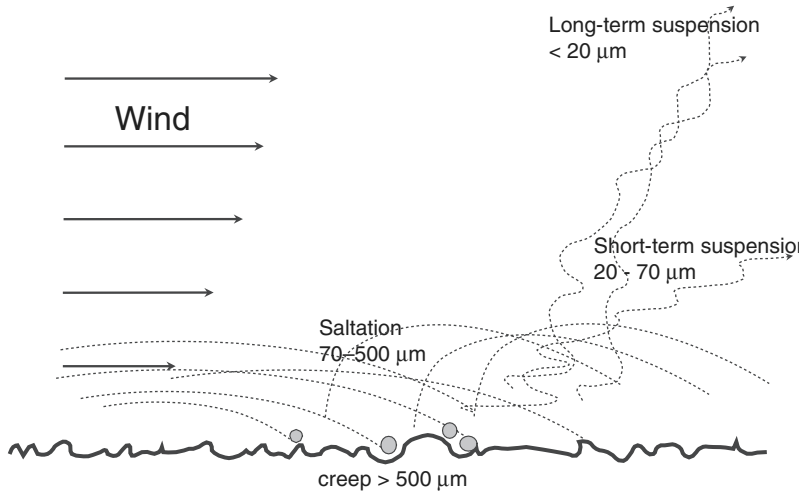

Figure 2.7.8 The three main transport modes of particles during wind erosion

Suspension. Particles with a smaller terminal velocity (u_F) than the vertical upward directed turbulent motions become suspended. The value of the vertical component of the wind in the boundary layer is approximately equal to the friction velocity u_*. Hence all grains with terminal velocities smaller than the actual friction velocity are transported upwards. For wind erosion events this is only relevant above the threshold friction velocity u_{*t}, so the boundary is (Greeley and Iverson 1985)

$$\frac{u_F}{u_{*t}} \Big/ \frac{u_*}{u_{*t}} = 1 \tag{2.7.11}$$

Furthermore, suspension can be divided into long- and short-term suspension. Particles smaller than 20 µm are subjected to long-term suspension, whereby they can be transported for several days over several hundred kilometres. Particles with diameters between 20 and 70 µm remain suspended for only a few hours and cannot be transported for very large distances (Shao, 2000).

2.7.4.1 The Saltation and Suspension Link

Saltation can be referred to as the driving process of wind erosion. Soil particles, which were lifted by fluid dynamic forces, accelerate in the higher layers and return to the ground. Owing to the momentum gained, these particles rebound and continue their movement in saltation and/or strike other grains. This causes a rapid increase in the downwind transport rate, also known as 'avalanching'. As the transport rate increases, the surface wind velocity decreases owing to the extraction of momentum by the grains in motion. After certain distances, equilibrium or the maximum transport capacity of the wind is reached. The transport capacity is independent of soil type and about the same for all soils, but the distance from the point of initiation to saturation varies with soil erodibility (Chepil, 1959). These distances differ between a few metres on highly erodible surfaces such as sandy beaches, where erodible material is not limited, to a few hundred metres on arable land, as shown in Figure 2.7.9. Aggregates or crusts at the soil surface generally reduce wind erosion on arable land by reducing the amount of erodible material. Therefore, the

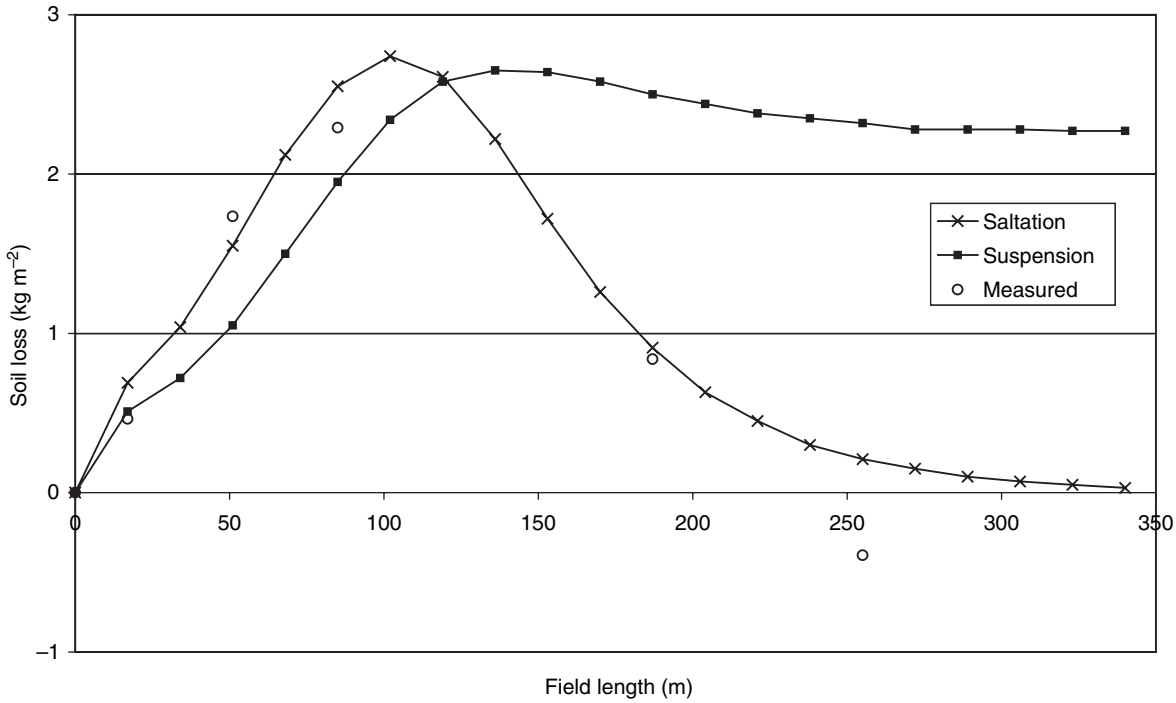

Figure 2.7.9 Soil loss over a field divided into the saltation and suspension part (measurements by Funk, 1995; saltation and suspension, calculated with WEPS)

transport is initially unsaturated. As long as the transport is unsaturated, the particles in the saltation layer acquire more energy by the acceleration in the higher layers than is absorbed, and this surplus energy in the saltation layer is responsible for the abrasion of aggregates or crusts and the production of more transportable material.

Another effect of the saltating grains is sandblasting. Very fine particles ($<$20 μm) can be ejected from the soil surface or from the aggregates, where they are generally embedded (Alfaro *et al*., 1997). Once detached, these particles stay in suspension even if the initial conditions of the threshold wind or friction velocity decrease (see Figure 2.7.3). Consequently, saltation is the determining process for the destruction of aggregates or crusts, the rapid increase in the transport intensity and the production of suspension-sized particles.

2.7.5 SUMMARY

Wind erosion is an important land surface process in Europe, which has acquired increasing relevance in the last decade. Especially the emission of the finest and most valuable soil particles has led to degradation processes affecting the agriculturally used areas and the atmosphere. The focus of consideration has changed from on-site (in most cases economic effects) to off-site effects (impacts on environment). Also, the discussions about the climate change and the associated effects on human society in general and agriculture in particular have set the problem of wind erosion in a broader perspective.

Significant progress has been made in the last 10 years. Computer and Geographic Information Systems (GIS) have improved the abilities for modelling and managing huge amounts of data. The political changes in eastern Europe and the Internet have facilitated and accelerated the exchange between scientists. Field and laboratory studies have been carried out at national and international levels, so that the knowledge about the mechanics of wind erosion has increased. Measuring devices have been developed or improved and reliable methods to quantify the windblown sediment applied. Wind tunnels are in use in Aarhus, Aberdeen, Bremen, Debrecen, Ghent, London, Loraine, Müncheberg and Paris to investigate the basic processes in the laboratory or at the field site.

Field studies have been carried out by Sterk (1997) in Niger and Funk (1995) in Germany to quantify the mass transport for single erosion events and to improve the understanding of the wind erosion processes. Studies with special emphasis on the dust fraction have been carried out by Herrmann (1996) in Niger and by Gross (2002) and Goossens and Gross (2002) in northern Germany. Aspects of erodibility, roughness and soil wetness to the erosion threshold and transport intensity have been investigated by Neemann (1991), Düwel (2000) and Cornelis *et al*. (2004). The combined effects of sand transport and dust production were investigated by Alfaro *et al*. (1997). Projects such as 'Wind Erosion on European Light Soils' (WEELS) and 'Wind Erosion and Loss of Soil Nutrients in Semiarid Spain' (WELSON) have promoted cooperation between European researchers.

The ascertainment of the soil losses by wind erosion and their detection seem closely related to developments of the measuring techniques. Especially laser technology has shifted the detection of particles to a new level in the last decade. Widely used methods also include real-time particle sizing in submicron dimensions and the vertical scanning of the atmosphere by Lidar. In spite of the progress in recent years, the transfer of the results to larger scales should be the next, European-wide challenge.

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2.8

Shallow Landsliding

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2.8.1 REPRESENTATIVENESS OF SHALLOW LANDSLIDES WITHIN EROSION STUDIES

Shallow landslides in upland watersheds can be a significant source of sediment, and may produce damage to on-site resources and to downslope areas. If a landslide reaches a stream, the landslide materials (1) may be transported far beyond the original depositional location if the sediment loading is consistent with the carrying capacity of the stream or (2) may create a sediment-chocked channel if the normal carrying capacity of the stream is exceeded (Lane *et al.*, 1988; Ward, 1994; Benda and Dunne, 1997a). The hazard associated to the second option can be high if dam-break failures occur, triggering debris flows or hyper-concentrated flows that can cause other damage, for instance on inhabited alluvial fans (Ermini and Casagli, 2003).

However, little is known about the spatial and temporal distribution of these mobilisable sediments and the processes by which they are generated and transported through the watersheds. The torrent sediment load can be derived from several supply processes, such as soil erosion, shallow or deep-seated landslides and riverbank collapses. Significant progress has been made in the understanding of the mechanisms of detachment and transport from soil erosion, but there is much less knowledge about landslides contributing to the sediment load (Richard, 1995; Oostwoud Wijdenes and Ergenzinger, 1999). Delivery of landslide materials is often a significant phenomenon, ignored or not fully accounted for in the erosion and landslide dynamics models suggested by the scientific community (Lane *et al.*, 1988; Benda and Cundy, 1990; Benda and Dunne, 1997b).

Modelling the delivery of landslides to streams necessitates determining where and when shallow landslids will occur, whether they are induced by rainfall or human activity, which volume will be mobilised and

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whether they will reach the stream or whether they will be stored as colluvium at the footslopes. There is a vast literature on this subject (Sidle *et al.*, 1985; Ward, 1994; Reid and Dunne, 1996) and on methods for documenting and computing the sediment supply to streams. Nevertheless, several physically based models have been developed by Bathurst (1991), Montgomery and Dietrich (1994), Wu and Siddle (1995), Dietrich *et al.* (1995), van Steijn (1996), Burton and Bathurst (1998) and Roering *et al.* (1999). These authors have made remarkable advances in proposing techniques and models to yield the spatial and temporal occurrence of landslides and the amount of material that can be delivered to a downslope channel. So far, research into the modelling of erosion-prone catchments (Bathurst *et al.*, 1998; Luckey *et al.*, 2000) is impeded by (1) insufficient detail representation of the sediment production and transport on the hillslopes, (2) the difficulty of accounting for their spatial variability (Mathys *et al.*, 2000) and (3) the rhythm of soil and regolith production (Oostwoud Wijdenes and Ergenzinger, 1999).

It is important, however, to estimate whether the sediment derives from hillslope source areas or from near-channel source areas. Several fieldwork studies suggest that the areas contributing sediment, representing less than 10% of a basin, are mostly the riverbanks (collapses), the steep slopes (overland flow with concentrated gully erosion, small hillslope debris flows) and the gentler slopes where the soil or the regolith is thickened by shallow landslides (Dietrich and Dunne, 1978; Plà Sentis, 1997). As evidence, the contributing areas may differ for each process, depending on predisposal and causal factors and also on slope morphology (Dietrich *et al.*, 1993).

This chapter defines the causes triggering landslides and discusses some classification criteria. The term 'shallow landslides', as opposed to 'deep-seated landslides', is then discussed. Each type and subtype of landslides are described in terms of mobilisable materials. Then the lines for research to characterise these shallow landslides and introduce them into process-based catchment models are considered. This is illustrated by research carried out on a gullied catchment of the south-east French Alps, where a method for characterising the susceptibility of hillslopes to shallow landslides has been applied.

2.8.2 SHALLOW LANDSLIDE TRIGGERS AND CLASSIFICATION CRITERIA

Landslides are ground failures driven by gravity, induced by natural triggering factors (abnormally high pluviometry, snow melt, seismic shock, river or sea undercutting) or human triggering factors (exploitation of water tables, mining) (Flageollet, 1988). Their shape and size vary because of the combination of several preparatory and triggering factors (dissolution, deformation and rupture by a static or dynamic load). They may be controlled by the topography (inclination and shape of the slope), the lithology (physical and geomechanical characteristics), the geological structure (dip, fault, discontinuity), the hillslope hydrology (pore pressures, water contents) or a combination of all these factors. Water intervenes in several ways. First, it acts by changing the state or consistency of materials (transition from a solid to a plastic and finally to a liquid state) (Coussot and Meunier, 1996); second, it lightens the terrain due to Archimedes thrust (Terzaghi's law) in relation to water table increases. The consequence for the slopes is a reduction in the inter-particle forces and the associated friction throughout the length of the rupture surfaces. Finally, water acts as an agent in the transport of materials.

This chapter considers exclusively shallow landslides affecting hillslopes and characterised by a lateral displacement. For landslides affecting plateaux or plains and characterised by a vertical displacement, the reader may refer to Flageollet (1988) and Leroueil (2001). Landslides are classified according to their mechanisms (movement types) and the nature of the displaced material (material type), together with information on their activity (state, distribution, style), i.e. the rate of development over a period of time (Varnes, 1978; Cruden and Varnes, 1996; Dikau *et al.*, 1996). Hungr *et al.* (2001) suggest a classification of flow-like landslides based on genetic and morphological aspects. The classifications sometimes take account of the velocity; this criterion has the advantage of indirectly expressing the risk level, which may or may not give people the opportunity to escape at the onset of the phenomenon. Two types can be distinguished

according to the landslide velocity. On the one hand, slow movements progressively deform and may be accompanied by ruptures, but most of the time with no sudden acceleration. These movements can be monitored and controlled and are not a direct threat to human safety. On the other hand, rapid movements may accelerate suddenly. They can be subdivided into two groups, depending on whether the material propagates as an intact or reworked medium.

2.8.3 SHALLOW LANDSLIDES VERSUS DEEP-SEATED LANDSLIDES

2.8.3.1 Concept of Shallow Landslides

Before describing the various types of landslide, the terms 'shallow landslide' and 'superficial landslide' have to be defined. Some authors attribute the term 'shallow' to a function of the thickness of the unstable mass. Varnes (1984) uses the term 'shallow' for landslides less than 2 m thick; those between 2 and 5 m are 'medium' and those over 5 m are 'deep-seated'; this thickness corresponds to the depth of the main rupture surface. This depth is difficult to estimate from the surface using solely morphological criteria and calls for specific investigation. This purely quantitative criterion seems very restrictive and it is better to define a shallow landslide as a function of the nature of the mobilised materials (pedological soil *stricto sensu*, superficial deposits, substratum), and the position of the potential rupture surface (for localised ruptures in contact with a structural, lithological or hydrogeological discontinuity such as at the superficial soil formation interface, the superficial substratum formation interface or, conversely, when the instability is located within a thick, loose formation).

Above a relatively arbitrary thickness threshold, these definition criteria have a further advantage: they take account of the texture of the moveable materials, either in mass or reworked, in terms of their capacity to hold water (bearing in mind that a shallow landslide will have a thickness of only a few metres and a relatively small volume, often less than 1000 m^3). The rheological behaviour, depending on the material's capacity to hold water, leads to a wide variety of phenomena, which may cover very variable distances (Leroueil *et al.*, 1996). This texture corresponds to three types of material chosen to describe and classify landslides, i.e. rock, debris (a predominance of coarse elements) and earth (predominantly fine elements). Superficial deposits affected by shallow landslides are weathered rock or glacial, periglacial, torrential or fluvial deposits. The main prone deposits are given for each type of shallow landslide described in Section 2.8.3.2 (slope deposits, loess, sands, silts, clays, moraines, pyroclastic deposits, volcanic ashes, pedological soils, etc.).

2.8.3.2 Landslide Types and Mechanisms

Table 2.8.1 gives the names of the main types of landslides as a function of the nature of the material mobilised. Five landslide types are specified: fall, topple, slide, spread and flow. These five types may sometimes be combined or may succeed each other, forming a sixth type, composite and complex movements, which consists of more than one type (e.g. a rotational–translational slide) or those where one type of failure develops into a second type (e.g. slump–earthflow). The latter type is not described, as it usually relates to deep-seated landslides.

2.8.3.2.1 Falls and Topples from Vertical Scarps

Falls and topples comprise free movement of material from steep slopes or cliffs. A topple is very similar to a fall in many respects, but normally involves a pivoting action rather than a complete separation at the base of the failure.

Their general characteristics are as follows: the shape of the rupture surface is usually smooth and vertical; the material falls suddenly from a main scarp following a preparation phase during which a slice of material is

TABLE 2.8.1 Landslide classification (modified from Dikau et al., 1996; Hungr et al., 2001).

Type	Rock[a]	Debris (20–80 % >2 mm) predominantly coarse[a]	Earth (80 % <2 mm) predominantly fine[a]
Fall	*Rock fall*	**Debris fall**	**Soil fall**
Topple	*Rock topple*	*Debris topple*	**Soil topple**
Slide	*Rock block slide* (translational)	*Rotational slide (slump)* **Debris slide** (translational)	*Rotational slide (slump)* **Slab slide** (translational) **Mudslide/slump–Earthflow**
Lateral spreading	Rock spreading	*Debris spreading*	*Soil spreading*
Flow	Rock flow (sackung)	*Debris flow* **Debris avalanche**	**Soil flow (mudflow)**

[a]Susceptibility to be 'shallow': **always**; *sometimes*; rarely or never.

separated, damaging the intact mass; and the volume and size of the fallen material are extremely variable, depending on the morpho-structural and lithological conditions of the slope. These phenomena occur on cliffs when the base is eroded by the action of the sea or of rivers. The falls are always sudden and very quick, whereas topples vary in speed from extremely slow to extremely quick, with acceleration and deceleration phases.

A fall starts with the detachment of rock or soil from a steep slope along a surface on which little or no shear displacement takes place (Figure 2.8.1a). The trajectory of the fallen material is rectilinear and vertical for slopes with an inclination of more than 70° (Cruden and Varnes, 1996). Material on slopes with an inclination between 45 and 70° moves by successive rebounds (Figure 2.8.1b), depending on the size of the material, the restitution coefficient and the angle between the slope and the trajectory of the falling mass. Throughout the length of slopes of less than 45° the falling material may roll. In the latter two cases, the material, which is very mobile, may move considerable distances from the source zone. Debris falls and soil falls are typically shallow landslides. Debris and soil falls are triggered in loose materials and their volume varies from a few to tens of cubic metres. The accumulated material, which is not consolidated, may be easily mobilised and transported.

A topple is the forward rotation of a mass of rock or soil about an axis located below the centre of gravity of the displaced mass (Figure 2.8.1c). Topples may lead to falls or slides of the displaced mass, depending on the geometry of the rupture surface and the orientation and extent of the kinematically active discontinuities. Topples are most frequent in coherent, tectonised and fractured rocks. Cruden and Varnes (1996) state that the way in which topples occur may be very varied: flexural toppling occurs in rocks with one preferred discontinuity system, oriented to present a rock slope with semi-continuous cantilever beams which may develop into retrogressive complex rock topple–rock fall (Figure 2.8.1d); block toppling occurs where the individual columns are divided by widely spaced joints; chevron toppling occurs along complex structural configuration, where the change of dip is concentrated at the surface of rupture to give a complex rock topple–rock slide (Figure 2.8.1e).

Falls and topples affect sedimentary (limestone, sandstone), eruptive (basalt, dolerite) or metamorphic (schist) rocks. In the last case, the exfoliation into small columns may provoke slow tilting, which gives rise to colluvial deposits. Loose earth types prone to topples include more or less compacted sand and clayey soils in which desiccation and humidification cracks may appear.

2.8.3.2.2 Slide

A slide is a mass movement of material throughout the length of a 'rupture' or 'sliding' surface. Slide could be rotational (the sliding surface is curved) or translational (the sliding surface is more or less straight). It depends

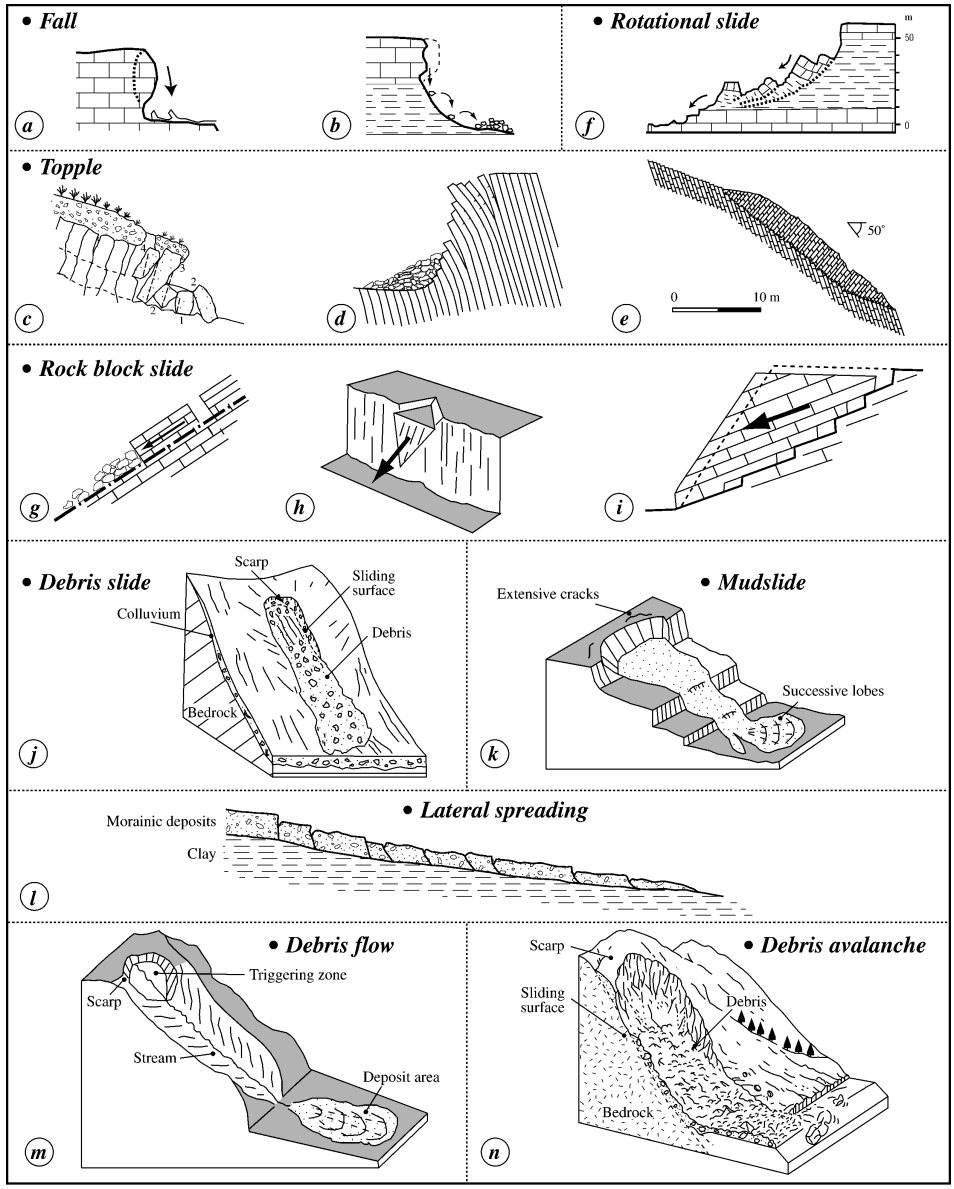

Figure 2.8.1 Typology of the main landslide types: (a, b) fall; (c–e) topple; (f) rotational slide; (g–i) rock block slide; (j) debris slide; (k) mudslide; (l) lateral spreading; (m) debris flow; (n) debris avalanche. (Adapted from Cruden and Varnes, 1996; Dikau *et al.*, 1996; Maquaire, 2005)

on the materials but also on the shape and length of the slope. A 1 : 10 ratio of depth to length is a criterion for the classification of a landslide, rotational or translational. Many slides are composite and the movement takes place over the length of a sliding surface which is concave upstream and flat downstream. Many slides also occur over an irregular surface (Flageollet, 1988), and they vary considerably because of the nature and size of the materials (fragments of coherent rock, loose rock, soil) and the velocity.

Figure 2.8.2 Successive and retrogressive translational slides in a moraine deposit covering a marl bedrock (Barcelonnette Basin, southern Alps, France, July 2002)

A single rotational slide is a 'more or less rotational movement, about an axis that is parallel to the slope contours, involving shear displacement (sliding) along a concavely upward-curving failure surface, which is visible or may reasonably be inferred' (Varnes, 1978). The morphology of the rotational slide is typical: upstream a main scarp with a steep slope, which is the visible part of the sliding surface, and tilted blocks (counterslopes) curtailed by scars along which slide striations are sometimes visible. They may be single, multiple, with several movements of the same type close to each other, or successive, i.e. interlocking (Figure 2.8.1f). Rotational slides can vary from terracettes with an area of only a few square metres (in this case, they are considered as shallow landslides) to large slides of several hectares. Generally, they occur in homogeneous loose formations (substratum or superficial deposits).

In translational slides (Figure 2.8.2), the material displaces along a planar or undulating surface of rupture, sliding out over the original ground surface. Translational slides often follow discontinuities more or less parallel to the slope, and are often superficial (such as contact between the rock and residual soils). Deeper, they occur along structural faults, joints or bedding planes.

Translational slides on single discontinuities in rock masses have been called rock block slides (Figure 2.8.1g) or planar slides (Cruden and Varnes, 1996). Sometimes the surface of rupture may be formed by two discontinuities that cause the contained rock mass to displace down the line of intersection of the discontinuities, forming a wedge slide (Figure 2.8.1h). A stepped slide may result if two or more sets of discontinuities, such as bedding surfaces and some joint sets, penetrate the rock masses (Figure 2.8.1i).This type of slide is usually very rapid. Smooth discontinuities or thin clay levels may act as a lubricant; infiltration water reduces friction, triggers excess pore pressures and provokes the sliding of one rigid block on another.

Translational slides can occur along soil–bedrock discontinuities or permeable–impermeable soil junctions in slopes formed by coherent, fine soils or coarser debris. In this case, translational slides are termed soil slides, debris slides (Figures 2.8.1j and 2.8.3) or slab slides. Debris slides and slab slides are normally shallow according to their length and width. Their velocities are linked to seasonal variations in groundwater levels and in the saturated conditions.

In some cases, a slide can change into a mudslide or slump–earthflow, especially on steep slopes, in highly tectonized clays or silty formations (Picarelli, 2001; Maquaire *et al.*, 2003). As shown in Figures 2.8.1k and 2.8.4, three morphological units can be distinguished: a primary source area with the main scarp and cracks, a

Figure 2.8.3 Shallow debris slide at the contact between slope deposits (thickness 3–5 m) and a marly bedrock (left bank of the Faucon stream, Barcelonnette Basin, southern Alps, France, July 2001)

Figure 2.8.4 Mudslide triggered near the town of Corps in the spring of 2001 (Trièves, southern Alps, France, May 2001)

track zone thinly covered by a flow (the paleotopography is still clearly discernible) and an accumulation zone, sometimes with several successive lobes.

2.8.3.2.3 Lateral Spreading

Lateral spreadings are lateral displacements situated at a certain depth and resting on stable formations. When they occur in coherent rock, whether or not resting on clay deposits or on plastic marl, they may affect a rock massif over considerable thicknesses and are therefore considered as deep-seated landslides. Further information can be found in Jahn (1964) and Bentley and Smalley (1984). We will confine our discussion to shallow lateral spreading occurring in non-cohesive heterogeneous materials such as moraine deposits (debris spreading) or fine clay or sand formations (soil spreading).

Fine formations particularly sensitive to this type of movement include varved clays deposited on the banks of the great Pleistocene glaciers in Scandinavia or over smaller areas in the old pro-glacier lakes of the Swiss Alpine border or in the Alpine valleys (Trièves in France; Nieuwenhuis, 1991). Their sensitivity is due to their very low plasticity. Sands may easily liquefy and give rise to this type of movement. Lateral spreading may also develop in coarse and heterogeneous moraine formations (Noverraz *et al.*, 2001).

These movements may occur on very gentle slopes of 5° or less. They display a characteristic morphology (Figure 2.8.1L): upstream at the source, a scarp with multiple lobes dominates the displaced part where horsts or rift valleys appear; downstream, these shapes are elongated by perpendicular folds in the direction of the movement. The space affected by these movements is often as wide as it is long and they often occur over a very short period of time.

2.8.3.2.4 Flows

A flow is a landslide in which the individual particles travel separately within a moving mass. They occur in highly fractured rock, clastic debris in a fine matrix or a simple, usually fine, grain size. Unlike slides, occurring along more or less well-defined shear zones, flow-like landslides are characterised by internal differential movements that are distributed throughout the mass (Picarelli, 2001). Their flow-like morphologies are much longer than they are wide and their uneven topography shows successive lobes. They have a considerable erosive capacity and can carry material eroded from slopes or banks over considerable distances and cover sizeable surfaces with varying thicknesses. They contribute to positive erosion balances, as the material can be carried outside the slope basin. Coussot (1993) suggests a rheological classification of these flows.

Hungr *et al.* (2001) distinguished several types of shallow flow-like landslides. Debris flow is a very rapid to extremely rapid flow ($>1\ \mathrm{m\ s^{-1}}$) of saturated non-plastic debris in a steep channel (Figure 2.8.1m). The key characteristic of a debris flow is the presence of an established channel or regular confined path, unlike debris avalanches, which are thin, partly or totally saturated, and which occur on hillslopes (Figure 2.8.1n; Malet *et al.*, 2003). Debris flows and debris avalanches are complex movements and usually start in the upper parts of slopes from slides (debris slide, slab slide) in loose, unconsolidated rocks and soil debris, especially where the vegetative cover has been removed by logging or fire (Ancey, 2002). They consist of a mixture of coarse material (gravel and boulders) embedded in a sandy–silty matrix, with a variable quantity of water (Costa and Wieczorek, 1987; Iverson *et al.*, 1997). In spite of varying velocities and total solid fractions, debris flows and debris avalanches show many similarities, particularly in the way in which they are triggered. Debris flows and debris avalanches are commonly triggered by an excess of water: intense rainfall, rapid snowmelt and, more rarely, glacier or lake overflows which mobilise unconsolidated material in their path. Rainfall intensity and duration, along with antecedent rainfall conditions, are strong controls for debris flow triggering (Dikau *et al.*, 1996).

Soil flows occur in wet sands or in silty clays which are so reworked with water or so liquefied by structural collapse that they adopt a flow mode (Cabrerra and Smalley, 1973; Locat and Demers, 1988; Locat and Leroueil, 1997; Hight *et al.*, 1998). A common term used for these conditions is mudflow. These are similar in form and behaviour to debris flows. They can be very slow to very mobile and can flow downslope fairly quickly. They tend to follow gullies or shallow depressions to spread out into a flat, bulbous fan or even a thin sheet. Soil flows may also occur in dry sand (dry sand flow); these are potentially very destructive but very rare. They originate when large masses of dry, non-cohesive, fine-grained material fall from steep slopes and fluidise on impact.

2.8.4 DELIVERY OF LANDSLIDE MATERIALS TO STREAMS: MODELLING STRATEGIES

After the description of the various types of landslide and their main morphological characteristics, the following questions must be addressed in order to determine the circulation of sediment in a landslide-erosion prone watershed: where are the slope failures located and what are the landslide runout distances?; how is material transported downslope? What is the linkage between the hillslope and the stream?; what is the percentage of landslide material deliveries to the stream (site factors, landslide morphology, material properties)?; what are the threshold conditions of the dynamics in the watershed (slope failure threshold, landslide material entrainment by the river). There is very little literature on the integrated modelling of catchment-scale erosion and subsequent sediment yield arising from shallow landslides (James, 1985; Bathurst, 1991). To answer the above questions, several process-based models should be used in conjunction and stepwise (Ward, 1994; Roehring *et al.*, 1999). To be of general practical use, the models should be physically based (yet parameter poor), so that they can be calibrated, however crudely, and to some degree validated.

The first step is to determine the location, the time and the volume of the landslides. The most developed GIS-driven models for predicting potential landslide occurrence at hillslopes are those analysing the threshold of slope stability, through a spatially explicit form, using digital topography, soil hydrology (in terms of steady-state rainfall required to saturate fully the soil mass) and planar infinite-slope stability models. Depending on the model used, output can vary from spatial distributions of steady-state rainfall predicted to cause slope failures (Reid and Iverson, 1992; Montgomery and Dietrich, 1994; Wu and Siddle, 1995; Pack *et al.*, 1999), through landslide-hazard potential based on factors of safety (Wu and Abdel-Latif, 1995; van Westen, 2000), to landslide-hazard rankings based on management criteria (Shaw and Vaugeois, 1999). More ambitious attempts at prediction are now incorporating calculations of finite-duration rainstorms and transient-state hydrology (Burton and Bathurst, 1994, 1998; Borga *et al.*, 1998; Dietrich *et al.*, 2001; van Beek, 2002) to estimate temporal magnitudes of hillslope failures and landslide-derived sediment fluxes. Model calibration requires the use of landslide inventory databases; hence the user may pick appropriate values for a specified catchment based on the ability of the resulting output to capture a high proportion of observed landslides and minimize the number of incorrectly identified sites (i.e. non-landslide points).

In a second step, yielding of landslide materials may be estimated by determining (1) the probability of a landslide reaching a stream with the help of GIS algorithms and (2) the rate of sediment deliveries through empirical studies. For instance, Ward (1994) proposed fitting logistic and linear regression models to some potential landslide characteristics (length, thickness, slope gradient) in order to estimate the volumes and rates of material the landslide would deliver. Because the delivered volume and rate are very imprecise values based on field and aerial photographic estimates, any estimates of yield have a very large degree of uncertainty. This empirical approach helps define key variables but does not explain the complex physical processes involved in the delivery of landslide material to a channel. Another option, therefore, is to link slope stability models to

landslide mobility models through physically based assumptions to estimate travel distance, velocity and debris deposit dimensions (Benda and Cundy, 1990; O'Brien *et al.*, 1993; Laigle and Coussot, 1997). Linkage of static slope stability models to dynamic mobility models is difficult (Bromhead, 1992). Dietrich *et al.* (1995) used a topographic model to predict shallow landslide initiation, runout and deposition zones on the basis of soil moisture, slope angle, soil density, angle of internal friction and slope gradient thresholds. More recently, Lenzi *et al.* (2003) and de Joode and van Steijn (2003) proposed physically based and GIS-driven models able to simulate landslide initiation, sediment propagation and deposits distribution.

Finally, in a third step, once the landslide material has been delivered to a stream, sediment transport functions (suspension load, bed material load) may be used to predict where sediment of a given particle size is likely to come to rest. The state-of-the art in sediment routing (O'Brien and Julien, 1997) indicates nevertheless that certain untested assumptions must be made about sediment mixing and the textural state of the channel bed (Richard, 1995; Julien, 1998). The conditions of the initial motion are fairly straightforward to specify and calibrate (Honda and Egashira, 1997; Montgomery and Buffington, 1997; Hubl and Steinwendtner, 2001).

Calibrating such complex catchment-scale models necessitates extended and adapted GIS databases for different watershed characteristics (topography, soil properties), rainfall thresholds and river discharges (Benda *et al.*, 1998). As evidence, collecting sufficient data in the fields can be problematic. Reasonable values potentially can be back-calculated by running the model with a range of possible values and choosing ones that yield landslide predictions most comparable to the observed locations (Carrara *et al.*, 1995). This approach might be less labour-intensive than field sampling but requires reliable landslide inventories in a sufficient number of representative watersheds so that the calibrated values can be extrapolated to basins without inventories. This calibration might inhibit the use of this model in watersheds with no viable analogues (e.g. geomorphologically similar watersheds with completed inventories). Moreover, given the influence of the topography on landslide locations, landslide runout tracks and erosion patterns, each digital-terrain based model is limited by the accuracy of the Digital Elevation Model (DEM) data; that is, these models are only as good as the DEMs on which they are based (Dunne, 1998). Data collection may also be complicated by the temporal evolution of the landslide controlling factors. For instance, landsliding resets the depths of soil and sediment accumulations and fires diminish root strength (Schmidt *et al.*, 2001). Moreover, unless the initial data on soil depths in landslide source zones, or some other field-based constraints, are entered into the model, most predictions will tend to overestimate the intensity of shallow landsliding because they fail to recognize sites that have been recently exhausted (Dietrich *et al.*, 1995). Surface wash and creep, causing deposition of colluvium at the footslopes, are also important process that should be considered in characterising the material distribution.

In order to give an insight into recognising ablation or accumulation sites in a watershed, Maquaire *et al.* (2002) proposed a methodology to estimate the soil depths potentially mobilisable and to locate landslide-prone areas on the basis of morphological signatures and soil hydro-mechanical properties. In the gullied watersheds of the Jurassic black marls of south-east France, shallow landslides (rock block slide, debris and slab slide; Figure 2.8.5) affect volumes from a few to several tens of cubic metres. These landslides can mobilise the weathered marl cover on the surface and in some cases the marl bedrock next to it. Over and above the constitution of a GIS-based landslide inventory, the initial input for modelling the delivery of landslide material to the downslope channels is the characterisation (nature, thickness) of the *in situ* regolith or colluvional superficial deposits, using resistance profiles obtained from regular drillings by dynamic penetrometry at variable energy levels (Figure 2.8.6a and b) and validation with observation pits. Some results are given in Figure 2.8.6c (Maquaire *et al.*, 2002), on which the thickness of the superficial deposit cover varies from 0.34 to 1.45 m, depending on the morphological situation (crest, slope, talweg). Digital terrain analysis and linear regression with GIS allow one to transpose the thicknesses on the basis of homogeneous topo-morphological facets.

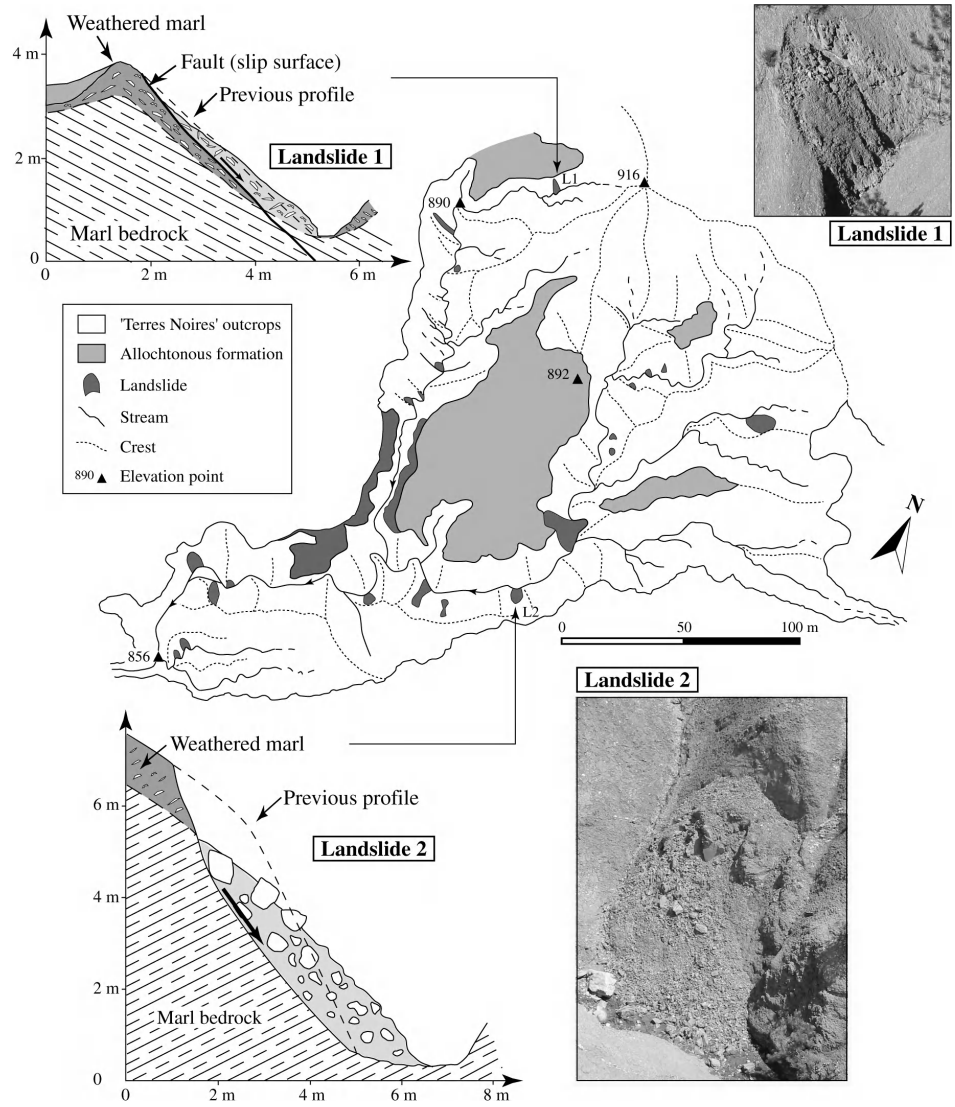

Figure 2.8.5 Inventory map of shallow landslides in the Moulin catchment (September 2002 to April 2003) and examples of two debris slides (Draix, southern Alps, France)

2.8.5 CONCLUSION

Shallow landslides are a major source of sediment delivered to streams. Individual landslides may mobilise in the form of a debris flow and subsequently travel several kilometres downstream, scouring channels of all sediment and wood. Although shallow landslides and associated debris or hyper-concentrated flows are an integral part of natural landscape processes, land-use management practices can greatly increase their spatial and temporal occurrence, which can lead to disturbance of stream habitat and loss of habitat features through high sediment loading.

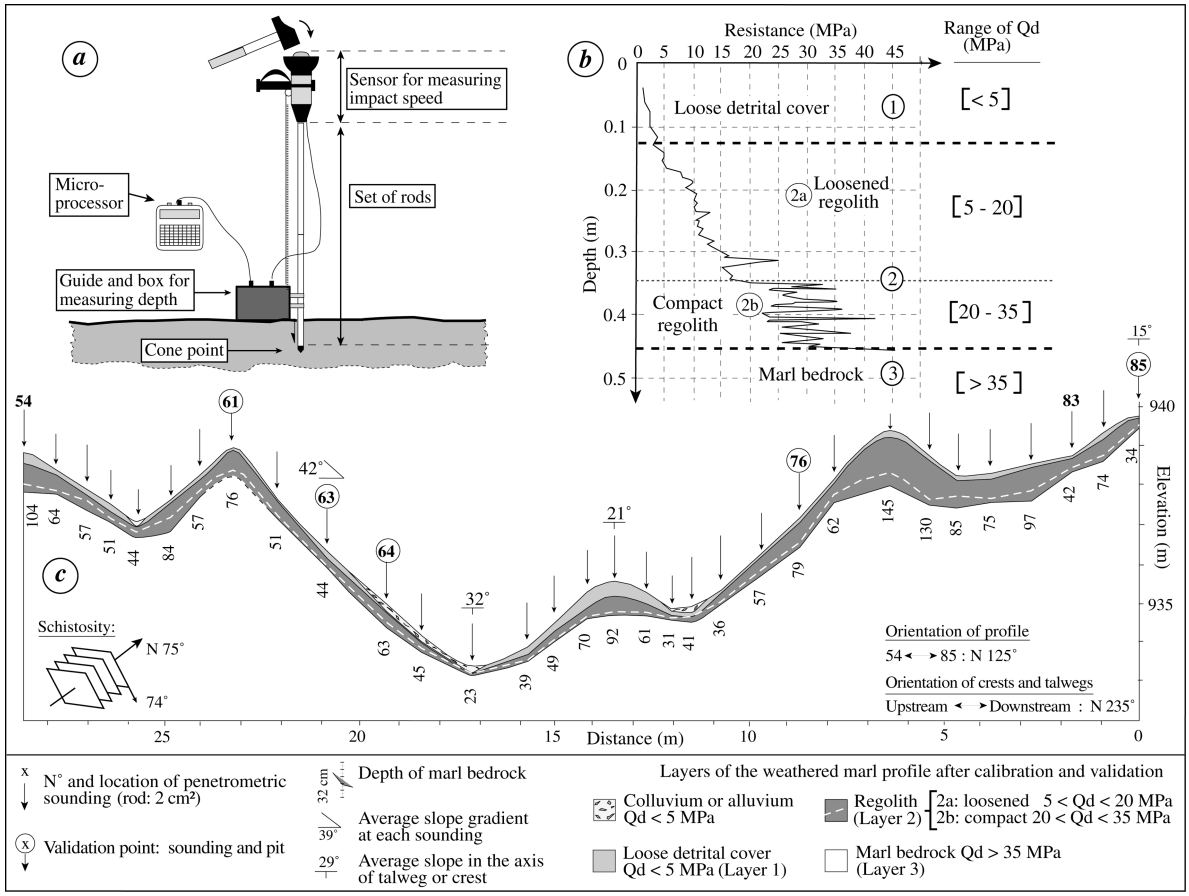
a
Sensor for measuring impact speed
Micro-processor
Set of rods
Guide and box for measuring depth
Cone point
b
Resistance (MPa)
Range of Qd (MPa)
Loose detrital cover
Loosened regolith
Compact regolith
Marl bedrock
[< 5]
[5 - 20]
[20 - 35]
[> 35]
Depth (m)
c
Schistosity:
N 75°
74°
Orientation of profile
54 ◂ ▸ 85 : N 125°
Orientation of crests and talwegs
Upstream ◂ ▸ Downstream : N 235°
Distance (m)
Elevation (m)
N° and location of penetrometric sounding (rod: 2 cm²)
Validation point: sounding and pit
Depth of marl bedrock
Average slope gradient at each sounding
Average slope in the axis of talweg or crest
Layers of the weathered marl profile after calibration and validation
Colluvium or alluvium Qd < 5 MPa
Regolith (Layer 2)
2a: loosened 5 < Qd < 20 MPa
2b: compact 20 < Qd < 35 MPa
Loose detrital cover Qd < 5 MPa (Layer 1)
Marl bedrock Qd > 35 MPa (Layer 3)

Figure 2.8.6 Weathering profiles obtained by dynamic penetrometry (Laval catchment, Draix, southern Alps, France): (a) description of the portable dynamic penetrometer with variable energy; (b) interpretation of penetrogram; (c) longitudinal cross-section with the type and thickness of the different layers of the weathered marl profile. (Adapted from Maquaire *et al.*, Copyright 2002, Elsevier)

Shallow landslides may occur locally and fairly suddenly, in different type of deposits, thus freeing considerable volumes of material. Nevertheless, catchment-scale erosion models take little account of them, first because the delivered volumes are underestimated compared with an areal erosion rate approximated through a measured load/contributive surface ratio from an experimental ablation station, and second because these phenomena are often misunderstood.

As indicated above, the general locations of landslide sediment sources are easy to recognize through field observation or GIS-based predictive modelling, but the intensity of the supply is a strongly nonlinear function of its driving variables (rainstorm size, slope gradient, soil strength, etc.), which calls for specialised research. Progress has to be made (1) in *in situ* soil hydro-mechanical characterisation (with non-destructive tools), (2) in the methods used for the automatic recognition of the morphological signatures of landslides from aerial or satellite photographs and (3) in the methods used for the automatic generation of specific detailed DEMs, at high frequencies and high resolutions, to estimate delivery rates after each landslide or flow event.

Multi-criteria landslide databases should be set up on several watersheds with different characteristics to calibrate and validate the models in use, although it is not yet clear how much initial data input or calibration procedures are needed to make them sufficiently accurate for operational use.

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2.9

Tillage Erosion

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2.9.1 THE PRINCIPLE OF TILLAGE EROSION

Whenever soil is cultivated, tillage translocation, which is the displacement of the cultivation layer, takes place. Experimental studies have demonstrated that the slope gradient has a dominant influence on soil translocation during tillage operations, as it is a gravity-driven process (Lindstrom *et al.*, 1992; Govers *et al.*, 1994, 1999). The net loss or accumulation of soil due to this spatial variation in soil translocation is referred to as tillage erosion. The term tillage erosion is often also used in a more general sense, indicating the process of soil redistribution in the landscape by tillage.

The basic nature of this process can be illustrated with up- and downslope tillage along a hillslope profile as an example (Figure 2.9.1). The unit soil transport rate in the direction of tillage, Q_s (kg m^{-1}), that is, the average net transport of soil per unit slope width due to a single tillage operation, can be calculated as

$$Q_s = \rho_b \bar{d} D \tag{2.9.1}$$

where ρ_b is the soil bulk density (kg m^{-3}), $\bar{d}$ is the average soil translocation distance in the direction of tillage (m) and D is the tillage depth (m). Tillage experiments have found mean translocation distances as a result of a single tillage operation to be linearly related to slope (e.g. Lindstrom *et al.*, 1992; Govers *et al.*, 1994; Poesen *et al.*, 1997; Quine *et al.*, 1999; Van Muysen *et al.*, 1999, 2000, 2002; Gerontidis *et al.*, 2001; de Alba, 2001; Van Muysen and Govers, 2002):

$$\bar{d} = a + bS \tag{2.9.2}$$

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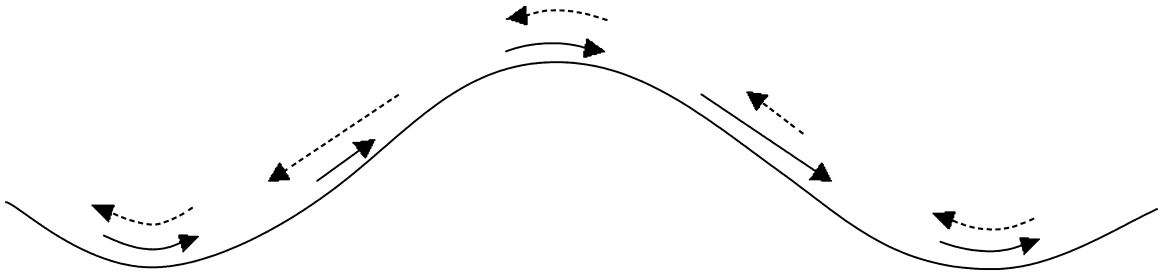

Figure 2.9.1 The principle of soil redistribution by tillage on a slope. Soil translocation is slope dependent and is smaller during an upslope tillage operation than during a downslope operation. This spatial variation in transport causes tillage erosion and deposition. The length of the arrow reflects the magnitude of soil translocation

where S is the slope tangent (positive upslope; negative downslope) and a and b are regression constants. Using Equations (2.9.1) and (2.9.2), and provided that the tillage direction is alternating up- and downslope, the average tillage erosion or accumulation rate over two consecutive operations, E (kg m^{-2}), may be written as (Govers *et al.*, 1994)

$$E = -\frac{\partial Q_s}{\partial x} = -D\rho_b b\frac{\partial S}{\partial x} = k_{til}\frac{\partial^2 h}{\partial x^2} \tag{2.9.3}$$

where h is the height at a given point of the hillslope and $k_{til}(= -D\rho_b b)$ is a constant. This means that the rate of tillage erosion may be characterised by (i) a single constant k_{til}, which is referred to as the tillage transport coefficient, and (ii) the rate of change in slope in the direction of tillage.

Evidence of tillage erosion is commonly observed as a difference in soil colour between hilltops and lower slope positions or as the development of soil banks near field boundaries. Tillage will cause soil loss on convexities such as crests and shoulder slopes and will cause accumulation in concavities, such as footslopes and hollows. Furthermore, parcel boundaries will act as a line of zero transport, so that soil loss will take place at upslope field boundaries, whereas accumulation will take place at downslope field borders, leading to the formation of lynchets or soil banks. It is worth noting that the spatial signatures of tillage erosion differ fundamentally from those of water erosion: soil loss by tillage will be most intense on landscape positions where water erosion is minimal (i.e. on convexities and near upslope field boundaries), whereas areas of soil accumulation by tillage are often areas where water erosion is maximal (i.e. hollows). This difference is illustrated in Figure 2.9.2.

2.9.2 QUANTIFYING TILLAGE EROSION

Tillage erosion is commonly studied by assessing the potential erosivity of a tillage implement, which is represented by the tillage transport coefficient (k_{til}). This value is determined by studying translocation of tracers on various landscape positions or is inferred from model simulations and can be used to quantify tillage erosion rates.

2.9.2.1 Experimental Measurement of Tillage Erosivity

In recent experimental studies, k_{til} values for various tillage implements and practices in Europe have been presented. For mouldboard tillage, k_{til} values for ploughing along the contour lines range between 48 and

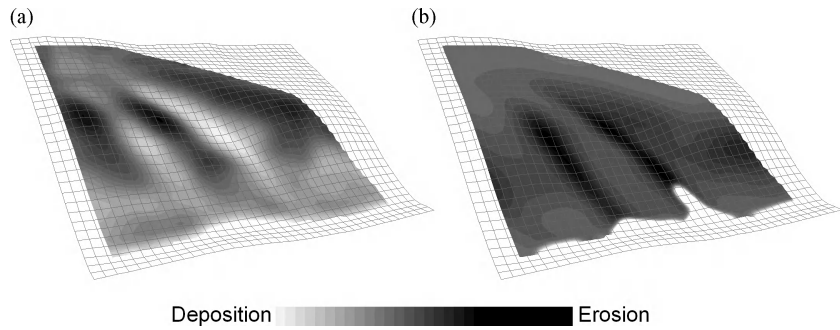

Figure 2.9.2 Typical spatial patterns of (a) tillage erosion and (b) water erosion simulated with the WaTEM model (Van Oost *et al.*, 2000a). The cell size is 6×6 m. The height difference between top and bottom of the field is approximately 12 m

360 kg m^{-1} per pass, whereas values for ploughing up- and downslope are usually higher and range between 65 and 769 kg m^{-1} per pass (Table 2.9.1). The former k_{til} values refer to soil fluxes based on translocation perpendicular to the tillage direction and the latter values to longitudinal fluxes in the direction of tillage. Experiments have shown that mouldboard tillage erosivity is affected by factors other than slope gradient alone, such as tillage speed, tillage depth and soil condition (Gerontidis *et al.*, 2001; Van Muysen *et al.*, 2002). Based on an extensive dataset, Van Muysen *et al.* (2002), proposed the following model for mouldboard tillage:

$$k_{long} = 2.026\rho_{\text{b}}D^{1.989}V^{0.406} \tag{2.9.4}$$

and

$$k_{lat} = 0.406\rho_{b}DV^{0.385} \tag{2.9.5}$$

where k_{long} is the tillage transport coefficient for an up- and downslope tillage operation (kg m^{-1} per tillage pass), V is the tillage speed (m s^{-1}) and k_{lat} is the tillage transport coefficient for a contour tillage operation (kg m^{-1} per tillage pass).

Van Muysen *et al*. (2002) suggest that mouldboard tillage erosivity increases exponentially with tillage depth for up- and downslope tillage, whereas the increase is linear for contour tillage. Tillage depth and speed are therefore important controls on tillage erosivity. Although studies have indicated that tool geometry affects soil translocation and erosion by mouldboard tillage, the effects have not been quantified so far.

Tillage transport coefficients for chisel/cultivator implements range between 70 and 657 kg m^{-1} per pass (Table 2.9.2). Chisel/cultivator erosivity is also influenced by tillage speed, depth, tool geometry and soil characteristics (Van Oost *et al.*, 2000c; Van Muysen *et al.*, 2000).

Significant tillage erosion is also observed during secondary tillage operations. Van Muysen and Govers (2002) report a k_{til} value of 123 kg m^{-1} for a rotary harrow and seeder where values between 9 and 333 kg m^{-1} have been observed for a disk harrow (Van Oost *et al.*, 2000c).

In intensive cropping systems, a whole sequence of tillage operations is required for crop cultivation. It is therefore useful not to express tillage erosivity on an implement basis, but to consider the erosivity of a typical sequence of tillage operations associated with common cropping systems. Govers *et al*. (1994) suggest that the values for the tillage transport coefficient, associated with individual tillage operations, may be summed to

TABLE 2.9.1 Tillage transport coefficients (k_{til}) for mouldboard tillage derived from tracer experiments for different tillage directions (up–down and contour), tillage speeds and tillage depths

Tillage depth (m)	Tillage speed(m s^{-1})	k_{til} (kg m^{-1} per pass)	Source	Country
Contour				
0.2	1.25	134	Gerontidis *et al.* (2001)	Greece
0.3	1.25	253	Gerontidis *et al.* (2001)	Greece
0.4	1.25	360	Gerontidis *et al.* (2001)	Greece
0.23	1.36	48	Heckrath *et al.* (2006)	Denmark
0.25	1.75	120	Heckrath *et al.* (2006)	Denmark
0.24	1.25	164	de Alba (2001)	Spain
0.26	1.35	184	Van Muysen *et al.* (2002b)	Belgium
Up–Down				
0.2	1.25	153	Gerontidis *et al.* (2001)	Greece
0.3	1.25	422	Gerontidis *et al.* (2001)	Greece
0.4	1.25	670	Gerontidis *et al.* (2001)	Greece
0.2	1.60	213	Van Oost *et al.* (2000c)	Spain
0.23	1.14	345	Heckrath *et al.* (2006)	Denmark
0.26	1.11	211	Heckrath *et al.* (2006)	Denmark
0.39	1.03	769	da Silva *et al.* (2004)	Portugal
0.15	0.75	70	Van Muysen *et al.* (1999)	Spain
0.33	0.51	254	Van Muysen *et al.* (1999)	Spain
0.18	1.25	65	Kosmas *et al.* (2001)	Greece
0.25	1.25	161	Kosmas *et al.* (2001)	Greece
0.21	1.64	98-104	Quine and Zhang (2004)	UK
0.28	1.25	234	Govers *et al.* (1994)	Belgium
0.27	1.8	263	Revel *et al.* (1993)	France
0.24	1.25	204	de Alba (2001)	Spain
0.25	1.38	224	Van Muysen *et al.* (2002)	Belgium
0.21	1.49	169	Van Muysen *et al.* (2002)	Belgium
0.21	1.74	177	Van Muysen *et al.* (2002)	Belgium

obtain average annual transport coefficients. Annual k_{til} values reported in the literature range between 400 and 800 kg m^{-1} yr^{-1} (Govers *et al.*, 1994; Van Oost *et al.*, 2000a).

It is noteworthy that most tillage experiments are conducted in, or perpendicular to, the direction of the steepest slope. Tillage direction is, however, not always determined by topography, but often by field geometry. Consequently, an important slope gradient may exist, both in and perpendicular to the direction of tillage. Recent experimental evidence has indicated that soil translocation in and perpendicular to the direction of tillage may not only be influenced by the slope gradient in the direction considered, but also by the complementary slope gradient (de Alba, 2001, 2003). Therefore, the effect of both these slope gradients on soil displacement should be considered when investigating the erosivity of a tillage implement.

2.9.2.2 Estimation of Tillage Erosivity from Caesium-137 (^{137}Cs)

This technique uses present-day ^{137}Cs inventories to optimise the parameters of spatially distributed soil erosion–deposition models that take into account all relevant processes (i.e. water erosion, tillage erosion and

TABLE 2.9.2 Tillage transport coefficients (k_{til}) for chisel/cultivator tillage derived from tracer experiments for different tillage directions (up–down and contour), tillage speeds and tillage depths

Tillage	Tillage depth (m)	Tillage speed ($m\ s^{-1}$)	k_{til} ($kg\ m^{-1}$ per pass)	Source	Country
Contour					
Chisel	0.14	0.65	139	Poesen *et al.* (1997)	Spain
Up–Down					
Chisel	0.15	1.25	111	Govers *et al.* (1994)	Belgium
Duckfoot chisel	0.16	0.65	282	Poesen *et al.* (1997)	Spain
Duckfoot chisel	0.19	0.7	605-657	Quine *et al.* (1999)	Spain
Chisel, stubble	0.15	1.57	169	Van Muysen *et al.* (2000)	Belgium
Chisel, pretilled	0.20	2.02	338	Van Muysen *et al.* (2000)	Belgium
Chisel	0.08	2.4	70	Van Oost *et al.* (2000c)	UK
Chisel	0.08	2.1	84	Van Oost *et al.* (2000c)	UK

soil loss due to crop harvesting), so that the observed ^{137}Cs redistribution pattern is predicted as accurately as possible (Walling and Quine, 1991; Quine, 1995, 1996, 1997, 1999; Govers *et al.*, 1996; Van Oost *et al.*, 2003a).

Table 2.9.3 presents k_{til} values derived from ^{137}Cs measurements. The clearest characteristic of the data is the high degree of similarity in the k_{til} values for various sites across Europe, ranging between 348 and 534 $kg\ m^{-1}\ yr^{-1}$. It is important to note that the k_{til} values derived from this technique represent average tillage erosion intensities over the last 35–45 years (depending on the sampling date) and are usually lower than present-day erosion intensities owing to the increase of mechanical power during recent decades.

2.9.2.3 Tillage Erosion Rates

While tillage transport coefficients allow comparison of potential tillage erosion intensity between tillage implements and management options, actual rates of tillage erosion are dependent on the interaction of tillage translocation with topography. Table 2.9.4 summarises typical tillage erosion rates on arable land in

TABLE 2.9.3 Tillage transport coefficients derived from ^{137}Cs data

Source	Country	k_{til} ($kg\ m^{-1}\ yr^{-1}$)	Time range (yr)
Govers *et al.* (1996)	UK	397	35
Govers *et al.* (1996)	UK	348	35
Van Oost *et al.* (2003a)	Belgium	523	38
Quine *et al.* (1997)	UK	550	35
Quine *et al.* (1996)	UK	300	35
Van Oost *et al.* (2000c)	UK	520	40
Van Oost *et al.* (2000c)	Denmark	500	40
Van Oost *et al.* (2000c)	Portugal	470	40
Van Oost *et al.* (2000c)	Italy	478	40
Van Oost *et al.* (2000c)	Greece	534	40
Van Oost *et al.* (2000c)	Spain	440	40

TABLE 2.9.4 Gross tillage erosion rates in Europe (eroded mass divided by the total surface area of the study area)

Source	Country	Tillage erosion rate (mg ha^{-1} yr^{-1})	Comments
Poesen *et al.* (1997)	Spain	24–93[a]	20 % slope transect
de Alba (2003)	Spain	16.7–67.7[a]	Field with complex topography
Govers *et al.* (1994)	Belgium	10	Regional estimate
Van Oost *et al.* (2003a)	Belgium	10.3	Field with complex topography
Van Muysen *et al.* (2000)	Belgium	9–19.5	0–20 % slope transect
Guiresse and Revel (1995)	France	3–16	Plot
Van Oost *et al.* (2000a)	Belgium	8.3–9.3[a]	730 ha catchment
Tsara *et al.* (2001)	Greece	4–18	

[a]Includes tillage erosion at field boundaries.

Europe inferred from modelling studies. These data, together with evidence from medium-term soil redistribution patterns derived from ^{137}Cs measurements and sequential aerial photographs (Govers *et al.*, 1996, 1999; Quine *et al.*, 1994, 1997), indicate that soil tillage contributes significantly to soil redistribution and landscape evolution on arable land in Europe. The erosion and deposition rates associated with present-day tillage techniques are such that, in many sloping agricultural landscapes, tillage erosion will be the dominant soil redistribution process (Vandaele *et al.*, 1996; Van Oost *et al.*, 2003a).

2.9.3 EFFECTS OF TILLAGE EROSION

Close relationships between the spatial distribution of tillage erosion and the spatial patterns of total C, N, P, texture, soil depth, rock fragment cover and above-ground biomass at various study sites across Europe have been reported (e.g. Poesen *et al.*, 1997; Schumacher *et al.*, 1999; Van Oost *et al.*, 2000a,c, 2003b; Kosmas *et al.*, 2001; Tsara *et al.*, 2001; Quine and Zhang, 2002). These results have provided evidence that tillage erosion operates like a conveyor belt, transferring soil and associated constituents from convexities to concavities. During cultivation, there is a net loss of plough soil from convex slope elements. However, the plough layer depth is maintained here by incorporation of nutrient-poor subsoil into the plough layer. Consequently, the plough soil on these eroded convexities becomes depleted in surface-applied or surface-immobilised nutrients and the products of weathering. This depleted plough soil is also translocated away from the convexities and, therefore, areas of no (or limited) net soil loss on linear slope elements below convexities may also be characterised by nutrient-depletion of the plough soil. Conversely, plough soil accumulates in concavities through downslope translocation from the upslope landscape elements. These relatively small areas, therefore, develop over-deepened plough soil enriched in nutrients. Therefore, translocation of soil by tillage erosion is a major contributor to within-field variability in soil properties. Model simulations indicate that continuing tillage will further increase the spatial variability of soil properties (Quine and Zhang, 2001; Van Oost *et al.*, 2003b) and has a deleterious impact on crop production (Schumacher *et al.*, 1999; Kosmas *et al.*, 2001; Tsara *et al.*, 2001).

It is therefore clear that tillage erosion should be considered when developing environmentally sustainable farming practices. First, attention must be given to tillage erosion in the development of strategies to manage soil variability. This is important when the value of precision farming needs to be addressed, as tillage will affect the spatial variation in soil properties. Second, tillage erosion effects on soil quality should also be considered in the debate about tillage strategies (conservation tillage versus conventional tillage).

2.9.4 MODELLING TILLAGE EROSION

2.9.4.1 Deterministic Tillage Soil Translocation Models

Torri and Borselli (2002) described the SETi model that simulates the 3D behaviour of translocated soil during tillage. This is a deterministic models that explicitly accounts for the interactions between tillage tool and soil. These interactions have been modelled as a three-phase motion: drag, when the soil is in contact with the instrument; jump, when the soil loses contact with the tool and is ejected; and rolling, when the clods roll and jump in relatively close contact with the soil surface. SETi can be used for calculating the tillage transport coefficient, and also for designing less erosive tillage equipment.

2.9.4.2 Tillage Erosion Models

Tillage erosion models may be used to simulate the loss or gain of topsoil. Govers *et al*. (1994) proposed to model tillage erosion as a diffusion-type process by calculating average translocation distances and soil fluxes between adjacent hillslope segments. This approach has been used in tillage erosion models for hillslopes (Govers *et al.*, 1994; TEP model, Lindstrom *et al.*, 2000) and for the 2D simulation of soil redistribution by tillage (WaTEM model, Van Oost *et al.*, 2000a). The assumptions of these diffusion-type models are that (i) tillage operations are performed in opposing directions and that (ii) soil translocation occurs in the direction of the steepest slope. More complex models of tillage erosion exist that take into account tillage direction and the interaction between complex topography and soil translocation (SORET model, de Alba, 2003; SPEROS model, Van Oost *et al.*, 2003a,b). These models provide excellent tools to assess the effectiveness of different management strategies.

2.9.4.3 Simulation of the Redistribution of Soil Constituents by Tillage

Tillage erosion not only results in the loss or gain of topsoil, but will also redistribute soil constituents within ploughed fields. Simulating the redistribution of soil constituents, however, requires a different modelling approach. Sibbesen (1986), Sibbesen and Andersen (1985), Sibbesen et al. (1985, 2000), Lobb and Kachanoski (1999) and Van Oost *et al*. (2000b) demonstrated that soil translocation is not uniform during tillage but that significant dispersion of soil takes place, i.e. not all soil particles are transported over the same distance (see Figure 2.9.3). Consequently, models that characterise tillage solely by means of the average translocation distance of the plough layer, which suggest uniform translocation, are not adequate for the examination of soil constituent redistribution within the plough layer. Van Oost *et al*. (2000b, 2003b) presented a tillage model that simulates soil profile evolution and the redistribution of soil constituents by convoluting the probability distribution of tillage translocation with the spatial distribution of the soil constituents.

2.9.5 CONCLUSION

Erosion and deposition rates associated with tillage reported in literature often exceed $10\,\mathrm{t\,ha^{-1}\,yr^{-1}}$, especially on fields with complex topography. Such rates are at least of the same order of magnitude as average water erosion rates reported for hilly cropland in western Europe. Consequently, tillage erosion should be considered as an important on-site soil degradation process.

Tillage erosion results in a soil redistribution pattern, which is, in contrast to the pattern of water erosion, controlled by the rate of change of slope gradient. This typically results in soil loss on the convexities, whereas deposition occurs in the concave landscape positions.

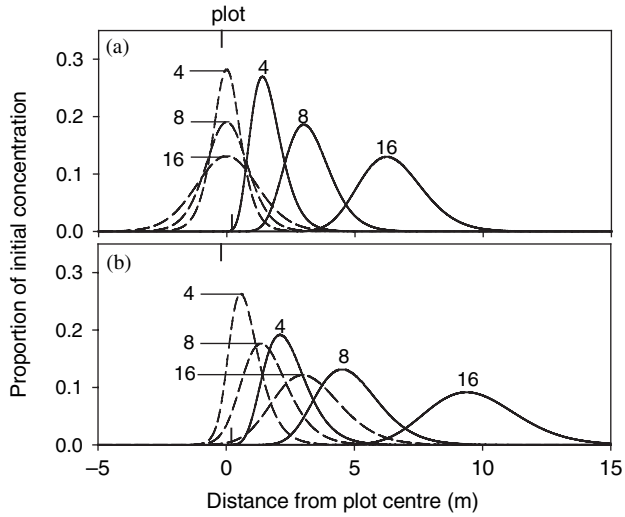

Figure 2.9.3 Illustration of soil dispersion by tillage. Dispersion and translocation of a labeled plot 0.4 m wide (a) on a flat surface and (b) on a slope of 0.22 m m^{-1} after n tillage passes in opposing directions (dotted line) or in one direction (solid line). (After Van Oost *et al*., *Soil Science Society of America Journal* 2000, **64**: 1733–1739. Reproduced by permission of the Soil Science Society of America)

The erosivity of a tillage implement can be characterised by a single number, the tillage transport coefficient (k_{til}). Values for k_{til} derived from tillage experiments range between 50 and 670 kg m^{-1} per mouldboard or chisel operation. The experimental data indicate that, in most cases, tillage erosivity may be reduced by

- reducing tillage speed and depth
- ploughing on consolidated instead of pre-tilled soils
- ploughing along the contour lines instead of up- and downslope.

Tillage erosion also has marked effects on soil quality: tillage will increase the spatial variation in soil properties and lead to a nutrient-depleted soil on convexities whereas a deep soil, enriched in nutrients, develops on concavities. Tillage simulation models can be used to assess the effects of tillage management practices on tillage erosion rates and soil profile evolution.

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2.10

Soil Losses due to Crop Harvesting in Europe

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2.10.1 INTRODUCTION

Growing crops may cause, in addition to water, wind and tillage erosion, significant soil losses at harvesting time. During the harvest, earth adhering to the crop, soil clods and rock fragments are removed from the field together with crops such as fodder and sugar beet (*Beta vulgaris* L.), carrot (*Daucus carota* L.), chicory (*Cichorium intybus* L.), potato (*Solanum tuberosum* L.), leek (*Allium porrum* L.) and black salsify (*Scorzonera hispanica* L.). This process of soil erosion is termed as soil losses due to crop harvesting (SLCH).

Although SLCH is often not included in soil erosion and sediment budget studies, some published data indicate that SLCH can be of the same order of magnitude as soil losses caused by water and tillage erosion (e.g., Auerswald and Schmidt, 1986; Belotserkovsky and Larionov, 1988; Eichler, 1994; Hasholt, 1983; Maier and Schwertmann, 1981; Poesen *et al*., 2001). SLCH may lead to considerable on-site loss of nutrients and valuable topsoil. Off-site, SLCH has economic and environmental consequences attributed to soil transport, cleaning of the crop and storage and disposal of the soil (Lindemans, 2002).

This chapter first focuses on methods to measure SLCH, SLCH definitions and the significance of SLCH. An overview of the factors determining SLCH is then given and, finally, spatial and temporal variations of SLCH in Europe are considered.

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2.10.2 SLCH MEASUREMENT METHODS

There are two main ways to estimate SLCH. The first method is by means of field measurements, whereby crop samples are taken during the harvest, preferably on the harvest machine itself. Subsequently, the crop is to be washed to assess SLCH. Field measurements have the advantage that the SLCH data can precisely be linked to determining factors such as soil texture and soil moisture content. A less time-consuming way to estimate SLCH is to use soil tare data from crop processing factories. In many factories, soil tare is measured in order to calculate the net crop mass of the delivery to be paid to the farmer. The advantage of factory soil tare data is that a regional or national average can easily be calculated. However, this method has some disadvantages. First, the mass of soil arriving at the processing factory is often not equal to the mass of soil lost from the field where the crop was grown (SLCH). This is due to post-harvesting practices such as crop storage and post-harvesting cleaning. Second, errors are introduced by assumptions made to calculate SLCH from tare data. Some sugar beet factories, for example, only measure total tare (soil, leaves, etc.), from which soil tare has to be estimated. Another example is that no soil moisture content data are available, which renders the calculation of the mass of oven-dry soil difficult. Third, data analysis using soil tare data is limited by poor or absent information about harvesting conditions and post-harvesting treatments.

2.10.3 SLCH DEFINITIONS

Ruysschaert *et al.* (2004) proposed some SLCH definitions, listed in Table 2.10.1, in order to allow comparison of SLCH data from different studies, often situated in the field of crop breeding, agronomy

TABLE 2.10.1 Definitions of SCLH and related parameters. (After Ruysschaert *et al.*, 2004)

Variable and units	Symbol	Formula[a]	Description
Soil losses due to crop harvesting	SLCH	—	Fine earth adhering to the crop, loose soil and rock fragments removed from the field together with the crop during the harvest
Mass-specific SLCH ($t\,t^{-1}$)	SLCHspec	$\frac{Mds + Mrf}{Mcrop}$	Soil loss value per unit of net crop mass
Crop-specific SLCH ($t\,ha^{-1}$ per harvest)	SLCHcrop	$SLCHspec \times Mcy$	Total soil loss for a given crop on an area-unit basis
Mean annual SLCH ($t\,ha^{-1}\,yr^{-1}$)	SLCHy	$\frac{\sum_{i=1}^{n}(SLCHcrop_i \times m_i)}{crc}$	Medium-term SLCH taking into account the crop rotation cycle
Net soil tare (%)	—	$\frac{Mds + Msm + Mrf}{Mcrop} \times 100$	Mass of non-plant material in a crop delivery to the factory, including soil, soil moisture and rock fragments, relative to the mass of the clean crop (net crop mass)
Gross soil tare (%)	—	$\frac{Mds + Msm + Mrf}{Mds + Msm + Mrf + Mcrop} \times 100$	Mass of non-plant material in a crop delivery to the factory, including soil, soil moisture and rock fragments, relative to the total mass of the crop delivery (gross crop mass)

[a]Mds = mass of exported oven-dry soil (t); Mrf = mass of rock fragments (t); Msm = mass of soil moisture (t); Mcrop = mass of the clean crop or net crop mass (t); Mcy = mass of the clean crop produced per hectare or net crop yield ($t\,ha^{-1}$ per harvest); n = number of different crop types grown in one crop rotation cycle ; m_i = number of times crop i is grown in one crop rotation cycle; crc = duration of the crop rotation cycle (yr).

and harvest machinery, and with soil loss data from other soil erosion processes. Different SLCH units are used at different temporal and spatial scales. A distinction is made between SLCHspec ($g\,g^{-1}$), SLCHcrop ($t\,ha^{-1}$ per harvest) and SLCHy ($t\,ha^{-1}\,yr^{-1}$). Whereas SLCH describes the intensity of the soil lost from the field where the crop was grown, the term soil tare is a measure of the mass of soil that is lost from the farm and is measured in crop processing factories. In the literature, soil tare is expressed as a percentage of net crop mass and also of gross mass of the crop delivery. To avoid confusion, a distinction is made between net and gross soil tare.

2.10.4 SIGNIFICANCE OF SLCH

Table 2.10.2 providing a literature overview of SLCHcrop values for several European countries, indicates that average values are in the order of several tons per hectare per harvest. In this table, three SLCH values are presented for sugar beet in Belgium. The first value ($8.7\,t\,ha^{-1}$ per harvest) for the period 1968–96 is given by Poesen *et al*. (2001). The second value for the period 1978–2000 (Ruysschaert *et al*., 2005) is based on the same data but completed with data from 1997–2000. The third SLCH value is calculated from data published in 1994 (Anonymous, 1994) and FAO crop yield data (FAO, 2003) and is opposed to the other SLCH values for sugar beet in Belgium not corrected for soil moisture content. The average SLCH values of all data found are $8.9\,t\,ha^{-1}$ per harvest for sugar beet, $1.7\,t\,ha^{-1}$ per harvest for potato, $8.1\,t\,ha^{-1}$ per harvest for inuline chicory, $11.9\,t\,ha^{-1}$ per harvest for witloof chicory and $6.8\,t\,ha^{-1}$ per harvest for black salsify. Maximum observed SLCH values from individual field plots may rise to a few tens of tons per hectare per harvest, e.g. $100\,t\,ha^{-1}$ per harvest for sugar beet, $33\,t\,ha^{-1}$ per harvest for potato, $71\,t\,ha^{-1}$ per harvest for witloof chicory, $66\,t\,ha^{-1}$ per harvest for carrot and $19\,t\,ha^{-1}$ per harvest for black salsify (Table 2.10.2).

Some estimates were made of the relative significance of SLCH in comparison with other soil erosion processes in a given region. According to Poesen *et al*. (2001), SLCH represents on average 19 % of the total mean soil loss in central Belgium. Depending on the region in Russia, SLCH may be as high as 33, 48 and 22 % of the water erosion in a given year for potatoes, sugar beet and carrot, respectively (Belotserkovsky and Larionov, 1988). Furthermore, Frost and Speirs (1996) stated that in 94 % of the area in East Lothian (Scotland), soil loss due to the harvest of crops such as potatoes and carrots, which was assessed to be $1\,t\,ha^{-1}$ per harvest, is larger than the soil loss due to water erosion caused by a severe storm with a return period of 20 years.

According to Hasholt (1983), the sugar beet harvest in Denmark causes a denudation rate of about $0.1\,mm\,yr^{-1}$. This is of the same order of magnitude as the findings of Eichler (1994), who found that the harvest of sugar beet in Baden-Württemberg, Germany, results in a denudation rate of $0.28\,mm\,yr^{-1}$. Verstraeten *et al*. (Chapter 1.30) calculated an average soil loss, caused by the harvest of sugar beet, potato, chicory root and carrot, in central Belgium of $1.8\,t\,ha^{-1}\,yr^{-1}$, which corresponds to a denudation rate of $0.13\,mm\,yr^{-1}$. Poesen *et al*. (2001) estimated for the same region a denudation rate caused by SLCH of $0.33\,mm\,yr^{-1}$. This value is probably somewhat too high, as Poesen *et al*. (2001) assumed that the average SLCHcrop value is $10\,t\,ha^{-1}$ per harvest and that a root or tuber crop is grown once every 2 years.

2.10.5 SLCH DETERMINING FACTORS

The factors determining SLCH rates are manifold. This section first provides an overview of the factors determining SLCHspec, then the factors determining the rate of SLCHcrop and SLCHy are discussed.

The factors determining the magnitude of SLCHspec can be grouped into four major categories, i.e. soil, harvesting technique, agronomic practices and crop characteristics, and are listed in Table 2.10.3. This table is based on a literature study of SLCHspec for sugar beet, but is expected to be similar for other crops. From this

TABLE 2.10.2 Literature review of estimated soil losses due to crop harvesting (SLCH) (After Ruysschaert *et al*., 2005, with permission from Elsevier)

Data type	Country/region	SLCHcrop (t ha^{-1} per harvest) [min-max][a]	Measurement period	Source
Sugar beet				
Field data	France[b]	14.0 [2.0–44.3i]	1984–86	Duval (1988), Poesen *et al*. (1999)
Factory data	Belgium	8.7 [4.4–19.5a][1–100i]	1968–96	Poesen *et al*. (2001)
	Belgium[c]	9.3 [4.7–19.4a]	1978–2000	Ruysschaert *et al*. 2005
	The Netherlands[c]	5.9 [3.4–9.8a]	1978–2000	Ruysschaert *et al*. 2005
	France[c]	13.8 [7.7–20.5a]	1978–2000	Ruysschaert *et al*. 2005
	Germany[c]	5.2 [2.2–10.7a]	1978–2000	Ruysschaert *et al*. 2005
	Germany/Bavaria	6.0 [2.9–9.1a][d]	1983–85	Auerswald and Schmidt (1986)
	Turkey	3.8[d]	1989–2000	Oruç and Güngör (2000), Oztas *et al*. (2002)
	Belgium	13.3[d]	1981–91	Anonymous (1994), FAO (2003)
	Denmark	10.4[d]	1981–91	Anonymous (1994), FAO (2003)
	France	16.9[d]	1981–91	Anonymous (1994), FAO (2003)
	Germany	8.0[d]	1981–91	Anonymous (1994), FAO (2003)
	United Kingdom	4.7[d]	1981–91	Anonymous (1994), FAO (2003)
	Italy	5.3[d]	1981–91	Anonymous (1994), FAO (2003)
	The Netherlands	9.3[d]	1981–91	Anonymous (1994), FAO (2003)
	Northern Spain	5.6[d]	1981–91	Anonymous (1994), FAO (2003)
Indirect estimates	Central Belgium[b]	$\leq$5.0 y	1956–87	Vanden Berghe and Gulinck (1987)
	Germany/Bavaria	15.0[d]	Not available	Maier and Schwertmann (1981)
	Mean	**8.9**		
Fodder beet				
Field data	Russia[b]	2.3[d][1.9–2.6i]	1985	Belotserkovsky and Larionov (1988)
Factory data	Russia[b]	3.5[d]	1985–1986	Belotserkovsky and Larionov (1988)
	Mean	**2.9**		
Potato				
Field data	Russia[b]	2.5 [1.8–3.4i][d]	1985	Belotserkovsky and Larionov (1988)
Factory data	Belgium[b]	2.1 [0.03 – 32.9i]	1999–2001	Biesmans (2002)
	Russia[b]	0.6 [0.1–1.1i][d]	1985–1986	Belotserkovsky and Larionov (1988)
	Mean	**1.7**		
Inuline chicory				
Factory data	Belgium	8.1 [3.2–12.7a]	1990–1996	Poesen *et al*. (2001)
Witloof chicory				
Field data	Belgium[b]	11.8 [1.7–70.5i]	1996–1997	Poesen *et al*. (2001)
Carrot				
Factory data	Belgium[b]	15.8 [0.5–65.5i]	1995–1996, 2001–2002	Van Esch (2003) and Soenens (1997)
	Russia[b]	1.3 [0.7–1.7i][d]	1985–1986	Belotserkovsky and Larionov (1988)
Black salsify				
Factory data	Belgium/The Netherlands[b]	6.8 [3.6–19.0i]	1995	Soenens (1997)
Radish				
Factory data	Russia[b]	1.7[d]	1986	Belotserkovsky and Larionov (1988)

[a] a = minimum and maximum values of annual averages for the country or region; i = minimum and maximum values at field plot scale; y = SLCHy (t ha^{-1} yr^{-1}) instead of SLCHcrop (see Table 2.10.1).

[b] Based on limited data.

[c] See also Figure 2.10.4.

[d] SLCHcrop = mass of oven-dry soil + mass of soil moisture instead of mass of oven-dry soil only.

TABLE 2.10.3 Factors determining SLCHspec for sugar beet grouped into four categories. (After Ruysschaert *et al*., 2005, with permission from Elsevier)

Category	Factor
Soil	Soil moisture content = *f*(weather, soil texture, soil structure)
	Soil texture
	Soil organic matter content
	$CaCO_3$ content
	Structure of the top soil
Harvesting technique	Type of harvesting machine
	Type of cleaning devices
	Length of cleaning circuit
	Type of lifting devices
	Harvesting operator
	Adjustment of the harvesting machine
	Harvesting depth
	Harvesting speed
	Adjustment of cleaning devices
	Equipment maintenance
Crop	Root shape
	Skin roughness
	Root grooves
	Rootlets
	Height of growth above the surface
	Root bifurcation = *f*(soil structure)
Agronomic practices	Inter row distance
	Plant density
	Crop yield
	Crop diseases
	Weeds and shoots
	Soil cultivation practice
	Crop homogeneity
	Sowing precision
	Seed bed homogeneity = *f*(soil cultivation)

literature review, Ruysschaert *et al*. (2004) concluded that soil texture and soil moisture content, influenced by soil properties and weather conditions at harvesting time, are key determining factors of SLCHspec. They were often found to be exponentially related to SLCHspec. The importance of soil texture and soil moisture content is illustrated in Figure 2.10.1 and although the original data set used for this figure is unknown, it provides a good overview of the relative importance of different SLCH-determining factors as experienced by the French sugar beet research centre (ITB). Other, less important, soil-related parameters determining SLCH are soil organic matter and $CaCO_3$ content (Duval, 1986, 1988) and structure of the topsoil. In addition to soil moisture content and soil texture, the category harvesting technique also has a major influence on the intensity of SLCHspec. Harvesting technique includes all factors related to harvesting machinery, such as type of cleaning devices, and all factors influenced by the harvest operator, such as adjustment of the cleaning devices and the harvesting speed. As these factors are human controlled and highly variable in space and time, harvesting technique induces large variability in SLCH, which is difficult to model. Variability is also induced, although to a lesser extent, by crop variety properties (e.g. skin roughness and height of growth above the

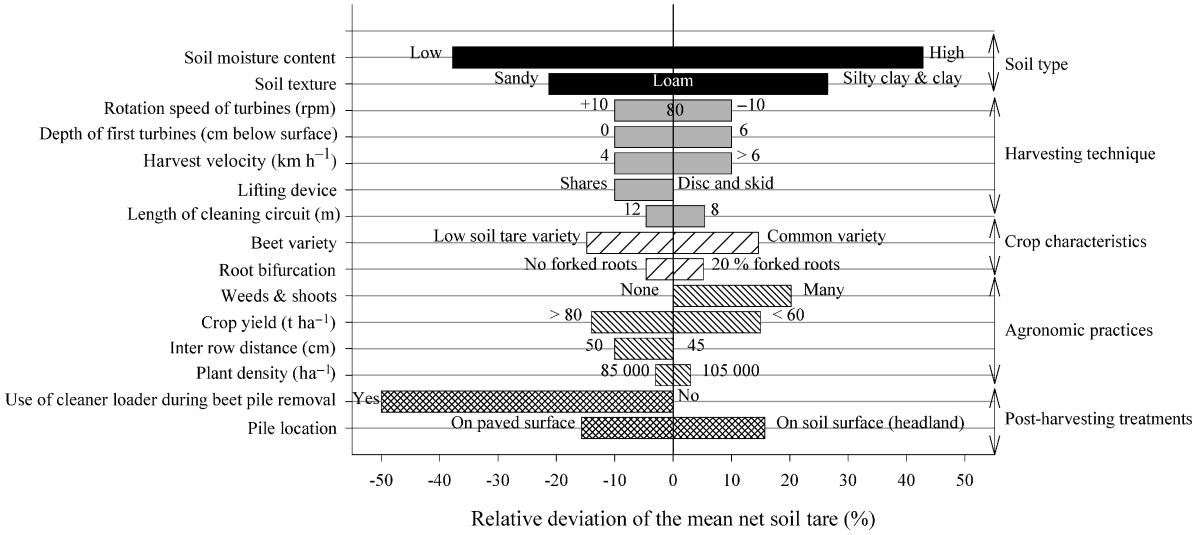

Figure 2.10.1 Relative change in soil tare of sugar beet due to a change of a determining factor. (After Ruysschaert *et al.*, 2004; Guiraud de Willot and Lévêque, 2002; ITB, 1999)

surface) and by the application of different agronomic practices. Only few applications of herbicides or none may, for example, lead to large quantities of weeds in the fields, which may block the harvesting machine, preventing proper cleaning of the crop.

As crop types (e.g. potato versus sugar beet) differ regarding harvesting technique, crop characteristics and agronomic practices applied, average SLCHspec values are expected to differ also.

Crop yield, in addition to SLCHspec, also determines the intensity of SLCHcrop (tons per hectare per harvest) (Table 2.10.1), which is illustrated in Figure 2.10.2. As a consequence, SLCHcrop is expected to be higher in highly productive areas compared with less productive areas.

In contrast to water and wind erosion, no correlation was found between SLCHspec or SLCHcrop and landscape position (Biesmans, 2002), although more research is needed to exclude topography as a determining factor of SLCH.

Finally, crop cycles have a major influence on SLCH on a medium-term basis, as is clear from the definition of mean annual SLCH (SLCHy) in Table 2.10.1. The more crops that induce SLCH (e.g. potato and sugar beet) are grown in a crop rotation cycle, the more important SLCHy will be.

2.10.6 VARIATIONS OF SLCH ACROSS EUROPE

It is clear that the spatial distribution of crops sensitive to SLCH is the dominant factor for the variability of SLCHy across Europe. Therefore, the most important crops that may lead to SLCH in Europe are grouped in Table 2.10.4. It can be seen that in 2000 those crops represented 14.3×10^6 ha or 4.9 % of the arable land of Europe and the Russian Federation. Of these 14.3×10^6 ha, 64 % is occupied by potatoes and 29 % by sugar beet. Potatoes (Figure 2.10.3) and, to a lesser extent, sugar beet (Anonymous, 2002) are widespread in Europe. Countries with a significant proportion of their arable land under potatoes are Malta (22 %), The Netherlands (19 %), Belarus (11 %), Iceland (10 %), Poland (9 %) and Belgium (8 %). Significant proportions of the arable land under sugar beet are found in The Netherlands (12 %), Belgium (11 %) and Slovenia (5 %) (derived

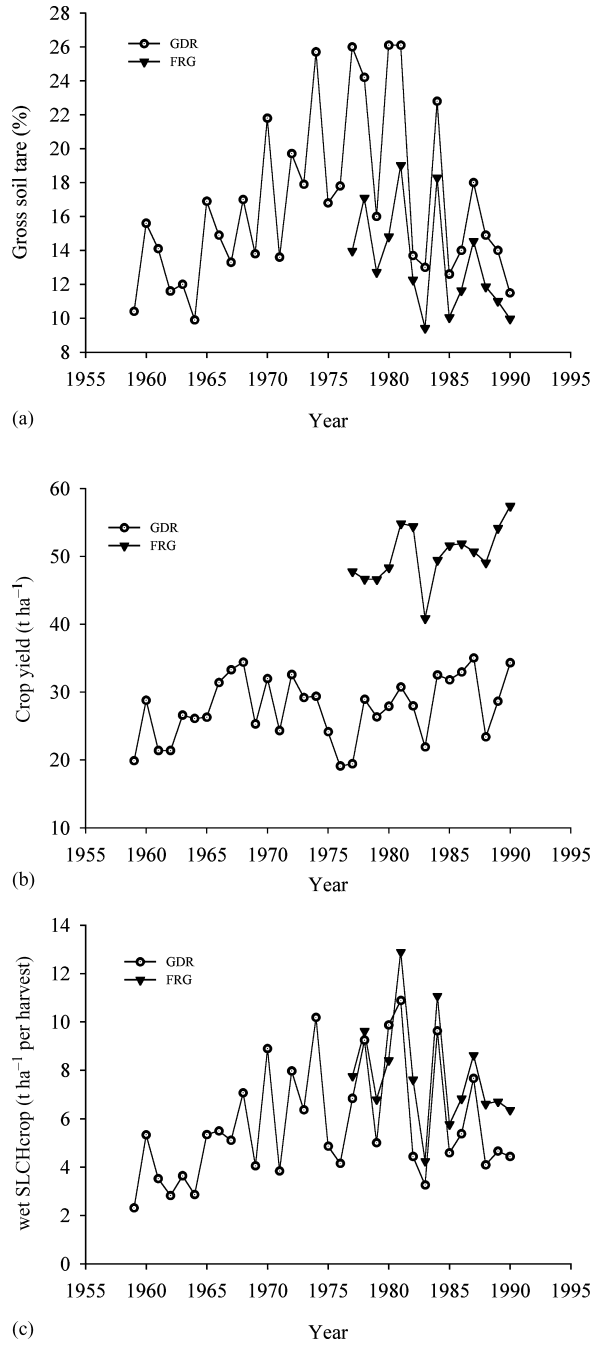

Figure 2.10.2 (a) Time series of mean annual gross soil tare for sugar beet grown in former West Germany (FRG) (source: WVZ, 1989, 1999) and East Germany (GDR) (source: Hessland, 1995). (b) Time series of mean annual sugar beet yield in former West Germany (FRG) and East Germany (GDR) (*source*: FAO, 1962, 1964, 1968, 1971, 1973, 1974, 1977, 1980, 1983, 1986, 1989, 1991). (c) Time series of mean annual soil losses (including soil moisture) due to sugar beet harvesting (wet SLCHcrop) in former West Germany (FRG) and East Germany (GDR), calculated from (a) and (b)

TABLE 2.10.4 Area (in 2000) in Europe and the Russian Federation vulnerable to SLCH, grouped per crop type. (*Source*: FAO, 2003)

Crop type	Area (ha)	Fraction of total SLCH area(%)	Fraction of total arable land(%)[a]
Potato	9 152 198	64.1	3.2
Sugar beet	4 164 689	29.2	1.4
Dry onion	431 848	3.0	0.1
Carrot	289 385	2.0	0.1
Garlic	128 541	0.9	0.04
Asparagus	57 027	0.4	0.02
Chicory root	21 265	0.1	0.007
Groundnut in shell	11 250	0.08	0.004
Green onion and shallot	10 277	0.07	0.004
Sweet potato	5 668	0.04	0.002
Other root and tuber crops	300	0.002	0.0001
Yam	130	0.0009	0.00005
Taro	100	0.0007	0.00004
Total area prone to SLCH	**14 272 678**	**100**	**4.9**
Total area arable land[a]	**289 729 000**		

[a]FAO definition: land under temporary crops (double-cropped areas are counted only once), temporary meadows for mowing or pasture, land under market and kitchen gardens and land temporarily fallow (less than five years).

Figure 2.10.3 Potato-growing areas in Europe. One major dot represents 1000 ha. (After Hijmans, 2001, with permission from AJPR)

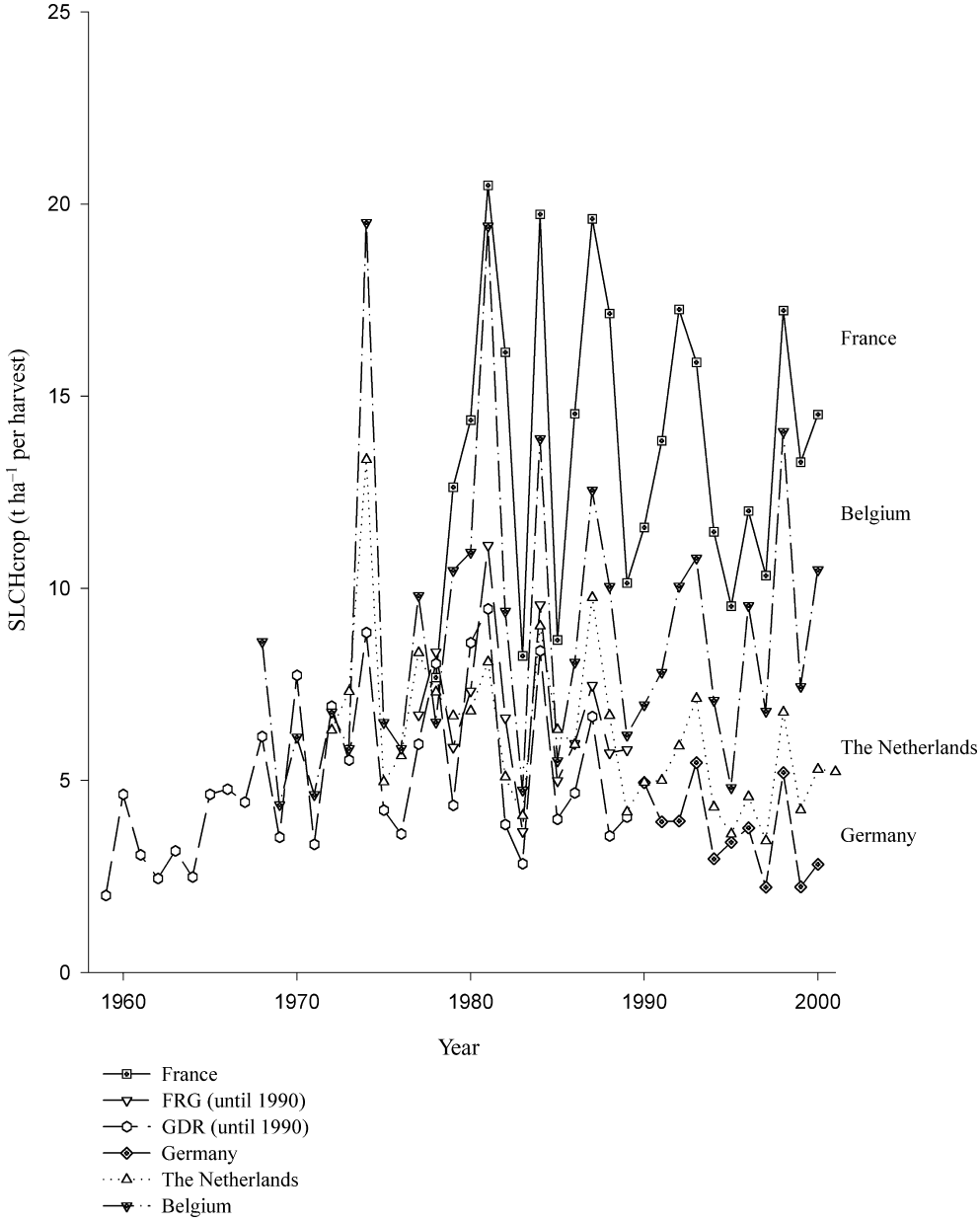

Figure 2.10.4 Time series of soil losses due to the harvest of sugar beet (SLCHcrop) for different European countries. Data are derived from soil tare measurements at beet-processing factories and from beet yield statistics and corrected for an assumed soil moisture content of 15 %. (After Ruysschaert *et al*., 2005, with permission from Elsevier)

from FAO, 2003). In some other countries, potato and sugar beet are regionally important, e.g., in northern France.

From Table 2.10.2 and Figure 2.10.4, it can be seen that large differences in SLCHcrop may occur between regions and countries. The rest of this section will focus on differences in SLCHcrop values for sugar beet.

SLCHcrop values for France are significantly higher than those for Belgium in the period between 1978 and 2000 (Figure 2.10.4). The values for The Netherlands and Germany are, moreover, significantly lower than those for France and Belgium (Ruysschaert *et al*., 2005). As soil moisture content at harvesting time is an important factor determining SLCH (see the next section), differences in rainfall depth recorded during the harvesting season could be a possible explanation for differences in SLCH between Belgium, France, The Netherlands and Germany. Rainfall depth of the harvesting season (September–December) appeared to be significantly less in Germany than in Belgium and France. However, rainfall could not fully explain these differences in SLCH due to the interaction between the effect of rainfall and country on SLCH (Ruysschaert *et al*., 2005). More research is needed to explore other possible causes for this variability. At present, only a list of possible factors affecting the variability of SLCH across Europe can be given (see the previous section). They are related to differences in soil characteristics, agronomic practices, harvesting techniques, crop variety and crop yield (Figure 2.10.2).

Crop yield appeared to be significantly higher in France than in The Netherlands and Germany during the period 1978–2000. However, this can only partly explain the higher SLCHcrop values in France as SLCHspec in this country is also higher than in Belgium, The Netherlands and Germany (Ruysschaert *et al*. 2005). Agronomic practices, harvesting technique and crop variety are human controlled and can be affected by the soil tare policy of crop processing factories, which in turn might be influenced by the environmental policy of the government. In countries where farmers receive large penalties if significant soil volumes are transported to the processing factory or where crop deliveries may be refused if the soil tare is too high, which is the case in, e.g., the UK and Sweden (Rigo, 2002), farmers might decide to apply more aggressive cleaning by the harvest machine and allow more beet damage and beet loss. However, soil tare policy affects the mass of soil entering the crop processing factories, but does not necessarily have an influence on the mass of soil leaving the field where the crop was grown (SLCH), if soil tare reduction occurred during post-harvesting practices. As a consequence, care has to be taken if SLCH is estimated from factory soil tare data, which was the case for the data presented in Figure 2.10.4 and many of the data in Table 2.10.2. Post-harvesting treatment practices such as post-harvesting cleaning occur in major parts of, e.g., Germany, the Czech Republic, Switzerland and the UK (Rigo, 2002).

Finally, differences in SLCH may be caused by different measuring systems and errors introduced when calculating SLCH from soil tare values. In some crop processing factories, sugar beet is, for example, dry cleaned before soil tare measurement. This dry cleaning is applied in, e.g., Italy and Greece and in some factories in Belgium, Germany and the Czech Republic (Rigo, 2002). Errors made during the calculation of SLCH from soil tare data may be due, to e.g., estimates of the soil moisture content at the time of soil tare measurement and uncertainties about the definitions of soil tare and crop yield. For the data for France (Figure 2.10.4), the error due to these uncertainties is a maximum of 2.5 t ha^{-1} per harvest (Ruysschaert *et al*., 2005). The large SLCH values for France in Figure 2.10.4 could be partly explained by large plant population numbers (number of plants per hectare), which cause larger beet surface areas to which soil can adhere, the use of a particular type of beet lifters on the harvest machine and the fact that farmers choose to have less beet loss, resulting in higher SLCH values (Anonymous, 1994). The latter reason may also explain why sugar beet yields are significantly higher in France than in Germany and The Netherlands.

2.10.7 INTERANNUAL VARIATIONS IN SLCHcrop VALUES FOR SELECTED COUNTRIES IN EUROPE

Interannual variations in SLCH for a given region (Figure 2.10.4 and summarised in Table 2.10.2) may have the same causes as variations between regions. However, the main determining parameter is soil moisture content during the harvest, which is influenced by weather conditions. This is illustrated in Figure 2.10.5, showing time series of average annual SLCHcrop values for sugar beet in Belgium and the rainfall depth

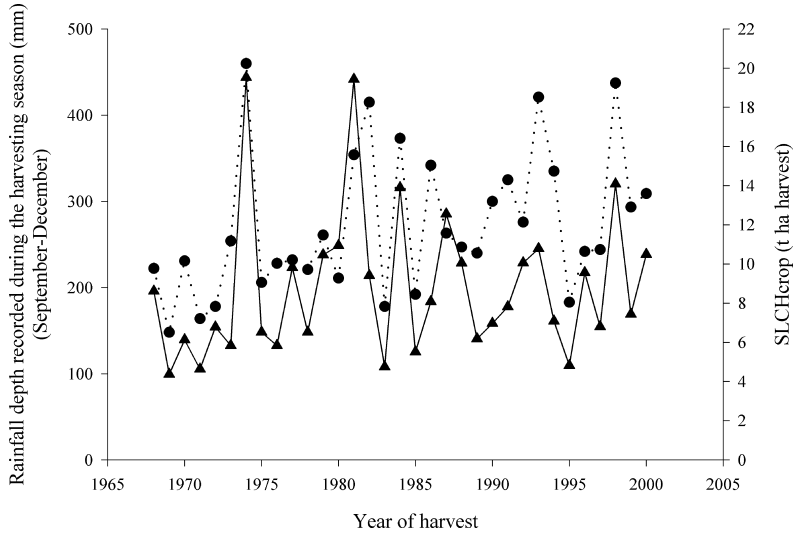

Figure 2.10.5 Time series of SLCHcrop values for sugar beet in Belgium (solid line) and the total rainfall depth recorded in Brussels (dotted line) during the harvesting season (September-December). SLCHcrop values are the same as in Figure 2.10.4. (Ruysschaert *et al*., 2005, with permission from Elsevier)

recorded during the harvesting season (September–December). Up to more than 50 % of the interannual variation of SLCHspec and SLCHcrop for sugar beet for the different countries indicated in Figure 2.10.4 can be explained by rainfall depth during the harvesting season (September–December) and by the year of harvesting (Ruysschaert *et al*., 2005). The latter is the consequence of efforts made by farmers and the sugar industry to reduce soil tare, which started from the end of the 1980s onwards. The decrease in SLCHcrop over time is often smaller than the decrease in SLCHspec as the decreasing effect is counteracted by increasing crop yield and could, in contrast to the decrease in SLCHspec, for most countries not be proven statistically (Ruysschaert *et al*., 2005). In France, the decline of SLCHspec can be attributed to improvements in harvest machinery and the fact that the industry and farmers pay more attention to the problem (ITB, 2001). Moreover, changes in post-harvesting treatments through time, such as cleaning, might also decrease the mass of soil arriving at the processing factories, which explains, among other reasons, the decrease in the case of, e.g., Germany (Schulze Lammers and Strätz, 2003). In 1990, 80 % of the sugar beet delivered to one German beet processing factory was cleaned after the harvest; for two other German companies, this value was about 5 and 20 %. In 2001, for all six German beet processing companies, the percentage of precleaned sugar beet deliveries was between 80 and 100 % (K Ziegler, Verband Fränkischer Zuckerrübenanbauer, Eibelstadt, personal communication, 2002). Post-harvesting treatments affect soil tare measured in the factory, but do not affect the loss of soil in the field where the crop was grown (SLCH).

2.10.8 CONCLUSIONS AND RESEARCH NEEDS

Crop harvesting may cause significant soil losses, especially in regions where several crops leading to SLCH are grown in the crop rotation cycle. These losses may range from several tons to a maximum value observed of several tens of tons per hectare per harvest. Assuming an average SLCHcrop value of 2 t ha^{-1} per harvest for potato and 9 t ha^{-1} per harvest for sugar beet (Table 2.10.2), the harvest of these crops, occupying 13 310 000 hectares or 93 % of the area sensitive to SLCH, is estimated to cause a total soil loss of 56 000 000 t yr^{-1} in

Europe and the Russian Federation. This implies that about 1 900 000 trucks are needed yearly to transport this soil from the farmer's field to storage places and crop processing factories! Given their significance, these soil losses should be incorporated in soil erosion and sediment budget studies. Spatial and temporal variations in SLCH for a given crop type may be influenced by many determining factors. An important determining factor of interannual variability of SLCH for sugar beet is rainfall depth during the harvesting season. However, owing to efforts by farmers and the processing industry, there has been a tendency for SLCH to decline in Western Europe, especially from the end of the 1980s onwards. As SLCH data for this study are derived from soil tare data measured in crop processing factories, more research is needed to establish where and when this decline took place. If this reduction occurred during harvest, more soil is left on the field, resulting in an effective decrease in SLCH. If this reduction occurred after the harvest, i.e. between the field and the soil tare measurement in the crop processing factory, this decline has no effect on the mass of soil lost from the field where the crop was grown. As research on SLCH as a soil erosion process is fairly recent, more research is also needed to increase insight into the importance of factors influencing spatial and temporal variability of SLCH.

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2.11

Erosion of Uncultivated Land

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2.11.1 INTRODUCTION

A large part of Europe is not cultivated. Indeed, 71.6 % of the former continent of Europe plus the states of Estonia, Latvia and Lithuania is not covered by arable land or permanent crops. About one-third of this 'new' Europe (FAO, 2000) has more than 10 % of its area covered by trees and is classed as forested (34.5 %; FAO, 2001), ranging by country from as low as 9.6 (Ireland), 10.6 (Denmark) and 11.6 % (UK) to 65.9 (Sweden) and 72.0 % (Finland); most countries have more than 25 % forest cover. Of the remaining non-cultivated land (37.1 %), much is grazed, lightly or intensively in unimproved and improved (fertilised) pastures, or above the tree line is very lightly or not grazed at all. Of the woodland, 90.1 % is considered 'natural' and the rest has recently been planted (FAO, 2001). Land above the tree line is also much less affected by mankind than are the grazed lands in the lowlands. Much of this uncultivated land is scenically attractive and has a greater biodiversity than the adjacent cultivated land. For these reasons, and because they are considered largely unspoilt and 'wild', these areas are important for recreation and tourism.

Uncultivated land includes a range of land types. These vary greatly and include the semi-arid landscapes of south-eastern Spain and Crete; the Mediterranean savannas, shrublands and forest; the forested slopes, meadows and rocky peaks of the alpine regions; the unenclosed rough grazings of the lowlands and uplands and peat moors of western Europe and Scandinavia; the wooded hills, mountains and plains of central and northern Europe; and the forest plantations found throughout these land types.

Much of the uncultivated land of Europe is considered not to be affected by erosion (Oldeman *et al.*, 1991), in places because it is thought to be 'non-used' wasteland. This is too simplistic a picture. Some of this land is being eroded, although erosion is often confined to particular localities. Such localities are often where mankind's impacts are, or have been recently, greatest. These impacts being driven by economic factors. However, our knowledge of erosion of uncultivated land is scanty for there are very few countrywide surveys

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of the extent of actual (not potential) erosion, those for Iceland (Arnalds *et al.*, 2001), England and Wales (McHugh *et al.*, 2002) and Scotland (Grieve *et al.*, 1995) being noteworthy.

Rarely is a distinction made between 'natural' erosion and 'accelerated erosion' (Grieve *et al.*, 1995), or it is considered that all erosion is human induced (Oldeman *et al.*, 1991) when it may not be, and it is accepted that any kind of erosion should be tackled (Fournier, 1972).

Erosion of uncultivated land is a much more complex topic, therefore, than is erosion of cultivated land because a wider variety of land types, land uses and erosional processes have to be addressed, and also, it will be argued, there are (mis)perceptions regarding erosion, its extent and severity. Some of this complexity will be addressed in the following sections.

In this chapter, the complexity of erosion of uncultivated land will be explored, erosion and erosion rates in different types of uncultivated land within Europe described and the more important causes of erosion discussed.

2.11.2 EROSION OF UNCULTIVATED LAND – A COMPLEX TOPIC

It is easy to see that land used to grow crops is greatly altered from its original state, especially in terms of its vegetation cover, and that its sensitivity to erosion is greatly enhanced because of this. Erosion processes act much more efficiently and vigorously on soils which are bare of vegetation for large parts of the year owing to cultivation. Under such circumstances, there can be little doubt that erosion has been accelerated when compared with that of the natural or, as in the case of most of Europe, the semi-natural state of the vegetated landscape.

It will be argued here that a distinction must be made, if possible, between 'natural' and 'accelerated' erosion, for it is only by making this distinction that a proper perspective on erosion can be gained. 'Natural' erosion rates can be high, for example in the Alps (Descroix and Gautier, 2002; Mathys *et al.*, 2003), and erosion of vegetated slopes by mass movements can be severe when large storms impinge on them (Starkel, 1976, Carling, 1986; Harvey, 1986), but neither of these can, arguably, be cost-effectively greatly alleviated by mankind's activities. However, erosion of arable and uncultivated land 'accelerated' by mankind's activities can be alleviated if there is a will to do so.

The perception of erosion is important. What appears to be severe erosion in the Mediterranean (McNeill, 1992) appears less so to other researchers (Grove and Rackham, 2001). Is the erosion of uncultivated land that we see now something that has recently begun, or is it something that has been happening for a long time? Thus, gullies can be spectacular but may have been recently initiated; for example, those in the wooded hill slopes in loess south of Lake Balaton in Hungary have formed in the last few decades (Kertesz, 2003), whereas spectacular gullies in badlands fringing the Mediterranean may have been eroding for over 4000 years (Wise *et al.*, 1982; Descroix and Gautier, 2002). In the first instance, erosion rates are high, or have been very recently, of the order of $100\,m^3\,ha^{-1}\,yr^{-1}$; in the second they are not, of the order of $5\,m^3\,ha^{-1}\,yr^{-1}$ or less (Canton *et al.*, 2001).

The last paragraph implies that we need to know what initiated the erosion of the uncultivated land that we see today. Is it something that happened recently or many decades or centuries ago? If recent, it is likely the causes can be identified, be it a large storm (Starkel, 1976; Carling, 1986; Harvey, 1986) overgrazing (Evans, 1977, 1996a) or burning (Anderson, 1986; Grove and Rackham, 2001). If in the past, it becomes much more difficult unless the history, especially the vegetational history, of the locality can be reconstructed, as in Iceland (Arnalds, 1992).

Another question is whether erosion is still occurring. It is in the badlands of the Mediterranean (Canton *et al.*, 2001; Grove and Rackham, 2001), or it may be in rough equilibrium whereby the bare soil that is being initiated is balanced by that which is being recolonised by vegetation as in the peat moorlands of Britain

(McHugh *et al.*, 2002). Or is the erosional form largely stabilised by vegetation and what we see now is mostly a relict feature, as seen in the eroding peat bogs of central and north-west Ireland (Large and Hamilton, 1991). In places, in the Alps and around the Mediterranean, on abandoned agricultural land or reafforested slopes, erosion is much reduced or alleviated (Descroix and Gautier, 2002; Mathys *et al.*, 2003).

There is also often an assumption that the shallow and stony soils that we find now in many parts of the uncultivated lands of Europe, especially the Mediterranean, are soils which have been severely eroded at some time in the past. This may not be so (Grove and Rackham, 2001). Soil development on hard rock is extremely slow. During the last Ice Age, which ended 10 000 years ago, in northern Europe and alpine areas, much soil was stripped from the land by glacial action or by solifluction, as it was again to a lesser extent in northern and western Europe during the Little Ice Age of the 14–18th centuries. Limestones, hard as in many parts of the Mediterranean, for example, or soft, as in the chalk of England, dissolve slowly and leave little residue. Other hard rocks are broken down physically by the weather and percolating water acts chemically on the fragments, but it takes a very long period for fine particles to be produced. In cool, humid western Europe it takes hundreds to thousands of years for organic residues to accumulate on rock to give shallow peat soils. In many localities, therefore, thin and often stony soils are not an indicator of past erosion.

The Universal Soil Loss Equation (USLE) (Wischmeir and Smith, 1978) and its successor empirical models are used for assessing water erosion risk. Such an assessment has been made for Europe (van der Knijff *et al.*, 2000), with the implication that such an assessment relates to actual erosion. This assumption is questionable. The erosion rate data used in the equation were or are collected from small plots. How these data relate to the landscape as a whole is unknown (Evans, 1993). Also, the USLE is used to estimate potential water erosion of uncultivated land, a task that it is not up to. Such a use implies that all the landscape is eroding, albeit at varying rates, and such a usage may possibly explain why many soil erosion researchers consider water erosion to be a 'natural' phenomenon. On well-vegetated slopes, sheet wash, that is, sheet erosion and small rills, the 'drivers' of the USLE, have no or little impact. Unless there is definite evidence of eroding bare soil on an uncultivated slope or, for example, signs of trampling and tracking by animals, it is best to consider such insignificant rates of erosion as the 'natural' rate of erosion.

What is considered to be erosion of uncultivated land is also important. The initiation of bare soil on slopes, for instance by animals, and its consequent erosion by the weather must count straightforwardly as 'accelerated' erosion, as would mass movements on deforested slopes. However, on slopes with little altered vegetation, mass movements would be considered 'natural' erosion. Should eroding river channels or channels extending headward be considered soil erosion? It is suggested here that they should be so considered as soil is being removed from the landscape, but in these instances a distinction between 'accelerated' and 'natural' erosion can be difficult to make.

Another point for consideration is closely related to the point made in the last paragraph and relates to stream channel erosion. Many researchers use sediment loads in rivers as a measure of soil erosion, a dubious assessment (Walling, 1983; Evans, 2002a). It is difficult to apportion where the sediments come from – slopes, channel sides or channel floors – and often high rates of sedimentation relate to high rates of 'natural' (or 'semi-natural') erosion and not to the implied 'accelerated' erosion. Besides, channels may be eroding rapidly, but that does not mean that the land between channels is eroding. Hence, although it may be sensible to equate rates of erosion of cultivated land measured as $m^3\ ha^{-1}\ yr^{-1}$ to rates of surface lowering in $mm\ ha^{-1}\ yr^{-1}$, it is not in those localities where sediment is largely derived from river channels and gullies.

A further point is that many erosion processes are at work on uncultivated slopes. Soil creep, governed by wetting and drying and freezing cycles, is a slow-acting process (Evans, 1974; Anderson and Cox, 1984; Auzet and Ambrose, 1996), except on bare soils or where animals form terracettes, and will not be discussed hereafter. Mass movements of many kinds can occur on forested slopes; they are caused by very rare storms (Orme, 1990). Deforestation lowers erosion thresholds (Crozier, 1986) and can trigger mass movements,

although large storms are often still needed to do this. Mass movements transfer material from slopes to channels, and the remnant bare soil surfaces are further eroded by the weather. Bare soil can also be initiated by grazing animals (Evans 1998); or as roads and tracks (Charman and Pollard, 1995; Froelich, 1995; Johnson and Nisbet, in press), pathways (Bayfield, 1973; Bryan, 1977; Liddle, 1997; Coleman, 1981) and skiing facilities (Bayfield, 1974; Watson, 1985; Descroix and Gautier, 2002) by various users and exploiters of the land; by fires caused by humans by accident or by lightening (Anderson, 1986; Grove and Rackham, 2001); drainage ditches excavated by foresters (Robinson, 1980; Francis and Taylor, 1989) and to improve the drainage of wet moors (Newson and Robinson, 1983; Robinson, 1985). On all these bare surfaces, the weather, especially rainfall at lower altitudes but also wind and frost action, and animals can act vigorously (Evans, 1997).

2.11.3 VEGETATION COVER OF UNCULTIVATED LAND AND OCCURRENCE OF EROSION

There is very little land in Europe that has not been affected by human activities. Even in the high mountains and the arctic north animals have been grazed where the land is suited, and often changed the vegetation. There is very little land in Europe, therefore, that has not been disturbed by humans. It is only in those areas that erosion can be truly considered to be 'natural'. Everywhere else, depending on the level of disturbance, erosion must be considered to be 'accelerated' by human activities. The greater the disturbance, the more likely it is that the resilience and sensitivity of the land will have changed for the worse.

Present-day vegetation cover in much of Europe may be very different from what it was before Neolithic times, about 4000 years ago, before humans and their animals occupied and used the land in sufficient numbers to change its vegetation cover. Since then, in many parts of Europe forests have been cleared (WCMC, 1997). In some places, clearance and overgrazing led to spectacular and still ongoing erosion, as in Iceland (Chapter 1.5; Arnalds *et al.*, 2001) In other localities vegetation cover has changed little in that period, for example, in the Mediterranean (Grove and Rackham, 2001); the greatest changes occurred in temperate Europe before 4000 BP as the climate changed after the last Ice Age. In the arctic zone of northern Europe vegetation has recolonised areas devoid of vegetation owing to the harsh climate in the Little Ice Age of the 14–18th centuries. In those areas where vegetation cover has hardly changed, what erosion occurs there may be considered 'natural' erosion. In some deforested areas, slopes have been reafforested without thought to the occurrence of erosion (see below).

Localities suffering badland erosion (Figure 2.11.1) are confined to particular soft and erodible rocks mostly around the Mediterranean, and are often associated with earthquake zones and tectonic movements. Such localities are inherently susceptible to water erosion, and small storms of 9 mm rainfall (Canton *et al.*, 2001), similar to those which trigger erosion of cultivated land (Evans, 1990a), can trigger erosion. Their unstable soil surfaces are inhospitable to plant growth, although protective lichens can be present. Hence even though erosion is continuing at present, and rates of erosion appear to be high (see below), it has been going on for a long time, resulting in the spectacular landscapes that we see now.

The greater part of the British Isles was tree-covered until about 4000 years ago (Evans 1996a). Since then, deforestation has taken place a number of times depending on waxing and waning population pressures, and grazing has inhibited recolonisation of trees in areas of land not enclosed for agriculture. Although thresholds of erosion have been lowered by forest clearance, large, rare storms, often of more than 70 mm rainfall (Newson, 1980, 1989) are still needed to trigger slope failures, although rainfall intensities to initiate mass movements are lower ($\sim$16 mm h^{-1}) than those which trigger widespread channel erosion ($\sim$49 mm h^{-1}). Intense storms can increase greatly the extent of channel widening and deepening, from about 3 % to over 20 % of channel length (Evans, 1996b). It can take many years for channels to stabilise.

Figure 2.11.1 Tabernas badlands, south-east Spain

In parts of the uplands of north-west Europe, tree clearance which changed the hydrology of the soils could also trigger the formation of peat. However, the more severe and wetter climate at higher altitudes or flanking exposed coasts as in south-west Ireland or the highlands and islands of north-west Scotland and Norway may have been sufficiently wet for peat growth to take place naturally. Peat is extremely vulnerable to erosion (Figure 2.11.2). Peat-covered slopes may become unstable because of the depth of peat formed on them (Glaser and Janssens, 1988). Large storms of 100 mm or more rainfall may trigger bog bursts and slides (Carling, 1986; Evans, 1996b), although smaller storms (25–50 mm rainfall) following dry conditions which cause desiccation of the peat and allow rainfall to enter the soil rapidly and cause liquefaction of the peat above an impermeable horizon can also trigger bog bursts (Alexander *et al.*, 1986; Wilson and Hegarty, 1993). Drainage ditches leading into the head of the burst or slide may also have lowered erosional thresholds (Wilson and Hegarty, 1993; Wilson *et al.*, 1996). Such events cause severe damage to river channels and biological systems (Crisp *et al.*, 1964; Wilson *et al.*, 1996).

However, although erosion of the peat may have been going on for a long time in the British Isles (McGee and Bradshaw, 1990), and could have been more severe in the Little Ice Age (Rhodes and Stevenson, 1997) and can be attributed to large storms (see the last paragraph), much erosion of the peat, and which is still occurring, was initiated by a combination of human activities and climate change (Tallis, 1997). Although some peat erosion in the Pennines started in medieval times, much, as elsewhere in Britain, started later in the middle of the 18th century. In both periods land pressure was great and slopes were cleared of their trees,

Figure 2.11.2 (a) Peat erosion in south-west Ireland and (b) 'Rofobard'-type erosion features in peat, Yorkshire Dales, England

climate was also drier and it is likely that the pools of water lying within the peat landscape dried out, exposing the peat. Desiccation may also have triggered piping (Jones, 1971; Burt and Holden, 2002), leading to gullying. In the 18th century, the industrial revolution produced moss-killing acid rainfall and in the 19th century, moors were burnt to help produce a crop of game birds which could be shot for recreational purposes. The changes in vegetation cover triggered erosion. In the last few decades, the moors in part have been overgrazed.

In mountain areas above the tree line, debris flows, snow avalanches (Van Asch and Van Steijn, 1991; Luckman, 1992) and other mass movements occur on steep slopes. Along with the relic solifluction lobes, they may have been more widespread at lower altitudes during the Little Ice Age. In this instance, such mass movements can be considered as 'natural' erosion. The removal of trees on steep slopes can trigger debris flows and similar mass movements, and these became widespread in Scotland in the 18th century and later (Innes, 1983) with the introduction of sheep and the use of fire. These mass movements can be considered as 'accelerated' erosion. In many parts of the Alps, forest clearance which triggered mass movements in the 18th and 19th centuries have been curtailed as slopes have been reafforested and climate has changed.

In many countries flanking the Mediterranean and in the Balkan states, former arable land, especially in isolated areas in the uplands, is being abandoned. Although instances of gully erosion occurring are reported, it seems likely that such instances may be isolated (Grove and Rackham, 2001) and may relate partly to the still high grazing pressures found in these localities. In many other places, trees and shrubs are reclaiming the former arable land, so reducing the risk of water erosion (MJ Haigh, Oxford Brookes University, personal communication).

Large gullies can occur in forested land, as they can on grass-covered slopes. Many of these are relict and overgrown and are decades or centuries old (Stankoviansky, 2003), although if overgrazed or a large storm occurs, parts of the gully system may be reactivated (Harvey, 1992). Active gullies, such as those in the rolling hills south of Lake Balaton, Hungary (Kertesz, 2003), are deeply cut into windblown deposits and it may be that such a substrate defines their occurrence. Thus, once incision started other processes (piping, vertical slips) characteristic of loessial soils came into play to result in these massive gullies. Such gullies may also be linked to runoff from arable fields above the forest. The wooded slopes are densely shaded for much of the growing season and often have a poor ground cover, and hence are vulnerable to runoff in spring/early summer from the cultivated fields above. Erosion stops when the gully floors become covered by 50–70 % of low vegetation (Rey, 2003).

In many upland areas of Europe, cleared land below the tree line or below the wall separating the steeper more rugged hills and mountains from the enclosed improved pastures is grazed intensively, as are the summer-grazed meadows above the forest in the Alps and the pastures in the wetter western parts of Europe. Although these pastures have been improved either by fertilisation or by long years of dunging by animals, they are rarely, if ever, ploughed. Such slopes have a strongly protective dense grass cover and rarely suffer water erosion except around field exits (Figure 2.11.3) and feeding points where animals congregate or on steep ($>15°$) banks. Shallow mass movements may occur in large storms. Such land is much less vulnerable to erosion than the unimproved rough grazings or cultivated land. Thus, improved grassland covers about 36.5 % of England and Wales but only accounts for 5.5 % of the land at moderate to high risk of erosion (Evans, 1996c), whereas unimproved grazings cover about 11.4 % of England and Wales but account for 21.9 % of the land at moderate to very high risk; figures for arable land are 44.5 and 72.6 %, respectively.

Much uncultivated land is scenically attractive and is attractive to visitors. Unfortunately, the soils, vegetation and climate of such localities often make the land vulnerable to erosion. Thus, coastal dunes and inland sandy heaths and also exposed uplands, especially scree slopes and peat moors, can all suffer severe erosion if too many people or vehicles travel across them (Liddle, 1997).

Figure 2.11.3 Trampled and eroding soil around gateway, Peak District, England

2.11.4 RATES OF EROSION OF UNCULTIVATED LAND

Mass movements can be massive in size, and in large slides or bog bursts many thousands of cubic metres can be transported downslope and downvalley. However, such movements often affect only very small parts of the landscape (Evans, 1993). Shallow landslides which may occur widely in some landscapes are discussed in greater detail in Chapter 2.8.

Within Europe there are some examples of field-based measurements of erosion rates of bare soil exposed within uncultivated land (Tallis, 1973, 1997; Evans, 1977, 1990b, 1996a; Phillips *et al.*, 1981; Birnie, 1993; Arnalds, 2000; McHugh *et al.*, 2002) and of the extent of erosion within a small catchment (Evans, 1977), a region or a nation (Phillips *et al.*, 1981; Evans, 1990c; Grieve *et al.*, 1995; Arnalds *et al.*, 2001). There are plot-based measurements of erosion for locations in some Mediterranean countries, as part of the MEDALUS project (Kosmas *et al.*, 1997; Grove and Rackham, 2001) and for badlands in the southern French Alps (Descroix and Gautier, 2002; Mathys *et al.*, 2003) and there is information on river sediment loads (Walling, 1988; Woodward, 1995), but such data apply to catchments which may contain both cultivated and uncultivated land, in addition to eroding river channels.

Rates of erosion of bare ground in the British uplands and Iceland vary greatly but can be up to many tens of m^3 or t ha^{-1} yr^{-1} (Evans, 1977; Arnolds, 2000; McHugh, 2000). Erosion faces can retreat by between 9 and 100 mm yr^{-1} (Evans, 1990b; Arnalds, 2000). Rates of annual surface lowering of peat (5–39 mm, Tallis, 1997; 1–40 mm yr^{-1}, Birnie, 1993) are similar to those measured on badlands in the southern French Alps (5–33 mm yr^{-1}, Descroix and Gautier, 2002), southern Italy (20–30 mm yr^{-1}, up to 70 mm, Alexander, 1982) and

southern Spain (6–18 mm yr^{-1}, Scoging, 1982) but higher than those from other badlands in southern Italy (3.5–10.5 mm yr^{-1}, after Rendell, and 0.6–0.8 mm yr^{-1}, after Servizio Idrografico Italiano, both quoted in Alexander, 1982) and from badlands in southern Spain (0.1–0.3 mm yr^{-1}, Canton *et al.*, 2001). Within badlands there can be 'hotspots' of erosion and rates of up to 258 t ha^{-1} yr^{-1} have been measured for small plots in the Ebro basin, north-east Spain (Sirvent *et al.*, 1997). In the Tabernas badlands of south-east Spain (Figure 2.11.1), rill channels cut down more rapidly than slopes are lowered in interrill areas (Canton *et al.*, 2001).

There is very little information on the rate of expansion of bare ground. Within countries it can be as much as tens (McHugh *et al.*, 2002) to hundreds of hectares per year (Arnalds, 2000). In Iceland, 39.7 % of the land is estimated to be considerably or severely eroding (Arnals *et al.*, 2001), whereas McHugh *et al.* (2002) estimate that only 2.5 % of the upland area of England and Wales is currently eroding.

Much of the material in temperate regions, unlike in badlands of the Mediterranean region, does not reach streams, as often large areas of bare soil do not flank rivers. Sediment can be delivered directly to streams where banks are eroding or mass movements occur on steep slopes adjacent to the channels. Sediment loads in the denser network of streams in upland England and Wales on average are higher than those in pastoral or arable areas (1.1, 0.4 and 0.2 t ha^{-1} yr^{-1}, respectively (Evans, 1996c) and can reach up to 2–3 t ha^{-1} yr^{-1} (Labadz *et al.*, 1991; Butcher *et al.*, 1993) on peat catchments in the Yorkshire Pennines. Similar figures have been reported for the Peak District (1.3 t ha^{-1} yr^{-1}, Hutchinson, 1995) and in Scotland on peat catchments (0.7–2.1 t ha^{-1} yr^{-1}, Duck and McManus, 1990). Such loads are similar to those measured in other upland areas in western and northern Europe (Walling, 1988).

Sediment loads in the French and Swiss Alpine rivers are often similar to those in northern and western European uplands (0.5–1 t ha^{-1} yr^{-1}, Walling, 1988), although they can be higher (up to $\sim$10 t ha^{-1} yr^{-1}, Descroix and Gautier, 2002), especially where channels are eroding (5.8–7.8 t ha^{-1} yr^{-1}, Becht, 1989), and particularly so where slopes adjacent to streams are actively eroding by mudflows. Thus material from only 12 ha of an 18.8-km^2 Alpine catchment increased mean sediment load from 5.8 to 7.4 t ha^{-1} yr^{-1}. Where alpine badlands occur erosion rates can be very high (100 t ha^{-1} yr^{-1}, Mathys *et al.*, 2003).

Stream sediment loads are generally higher elsewhere in southern European mountains (1–5 t ha^{-1} yr^{-1}, Walling, 1988). In the western Mediterranean, however, sediment loads are much higher, more than 5 t ha^{-1} yr^{-1} (Walling, 1988) and up to 19–42 t ha^{-1} yr^{-1} (Woodward, 1995). The variability in annual suspended sediment load can be high from a small badland catchment in southern Italy, ranging from 1.8 to 101.3 t ha^{-1} yr^{-1}, with a mean of 20.8 t ha^{-1} yr^{-1} (Porto *et al.*, 2001) These high loads are associated with tectonic instability, high relief and the presence of erodible soft rocks (badlands). Sediment delivery to streams is high, partly because some of the land in these catchments is cultivated, but probably mainly because channels and adjacent slopes are eroding. In some instances this may be attributable to rapid runoff from overgrazed slopes (see below).

In Britain, sediment loads may be higher for a time in afforested or ditched and drained moorland catchments compared with adjacent undrained moors until plough furrows and ditches become stabilised by vegetation. Sediment loads may be up to 16 times higher (Soutar, 1989), but more commonly 3–4 times (NCC, 1989).

As noted above, erosion of bare soil in uncultivated land can be high, on a par with erosion of arable land (Evans, 2002b), but whereas arable land may be kept bare of crop for much of the year, and every year, on uncultivated land vegetation will recolonise and stabilise the bare soil (Grove and Rackham, 2001; Evans, 2005) unless this process is hampered in some way, by animals for example (see below), or because the climate is severe. Inherently, therefore, if left undisturbed most bare soil on uncultivated land, however created, will become over time protected from erosion by a succession of lichens, mosses and then higher plants, and erosion rates will diminish.

2.11.5 GRAZING OF UNCULTIVATED LAND

Much of the uncultivated land of Europe is grazed. Sheep and goats leave obvious tracks on the slopes of semi-arid south-east Spain and their presence is recorded on many photographs taken of Mediterranean lands (for example, see Grove and Rackham, 2001). In such environments, sheep move rock fragments downslope by trampling (Oostwoud Wijdenes *et al.*, 2001). Sheep initiate bare soil in the Swiss Alps and the trampling of slopes by cattle is also not difficult to discern when walking through parts of the Swiss and German Alps. Intensive grazing of improved pastures by dairy herds in Britain creates bare soil around gates and along tracks, all of which can erode and soil particles are carried into rivers (Figure 2.11.3). Also in Britain sheep and beef cattle on lowland heaths or upland moors can create much bare soil around feeding stations. Wherever animals are herded in large numbers they are likely to break down river channel banks and initiate gullies down to the water (Figure 2.11.4).

Too many sheep on upland slopes can easily break down the fragile vegetation cover and initiate bare soil (Evans, 1997, 2001). In Britain, one sheep or lamb per hectare per year or one sheep per 0.2–0.3 ha in summer will trigger erosion of mineral soils. Erosion of peat is initiated at much lower (about 10 times) grazing intensities. Sheep constantly disturb the soil surface and impede recolonisation by vegetation, and the action of wind, water and frost can lead to rapid expansion of bare soil. A reduction in grazing intensity may allow the soils to be recolonised but it may take many years before that happens completely. In the Peak District in England, a reduction in grazing pressure of 30 % has been sufficient to allow recolonisation of mineral soils by vegetation (Evans, 2005). However, where peat occurs, it may be that not until all the peat has eroded, plus the underlying very leached and acid mineral soil horizon below, will vegetation be able to take a hold and erosion cease.

Figure 2.11.4 River banks broken down by cattle, Gwent, South Wales

Figure 2.11.5 Bare soil exposed along fence line by trampling by reindeer, Finnmark, Norway

The impacts of animals are most vividly seen in Iceland (Chapter 1.5; Arnalds, 2000; Arnalds *et al.*, 2001). Here, forest clearance from medieval times onwards plus grazing sheep and now horses (Arnalds, 1992) and a harsh climate have initiated much bare soil on the highly erodible volcanic soils. Wind erosion became severe and it is very difficult to stabilise such soils (Figure 2.11.2). It is estimated (Arnalds *et al.,* 2001) that on 51.6 % of Iceland's useable grazing lands erosion is sufficiently severe that grazing pressures should be reduced, and that no grazing should be allowed on 43.4 % of this land.

In semi-arid Finnmark, northern Norway, reindeer have trampled and tracked slopes and river banks, caused gullying, and along fence lines initiated massive damage to low vegetation, so creating bare soil (Figure 2.11.5) (Evans, 1996a). In this harsh climate, even removing animals may not encourage lichens and vegetation to recolonise the bare soils for many years. It is noteworthy that in other parts of northern Norway, where neither reindeer nor sheep graze the fells in large numbers, only occasionally are bare patches of soil found.

In both Norway and Britain, a doubling of the numbers of animals grazing the fells was sufficient to initiate and maintain erosion (Evans, 1998), as it was in Connemara, western Ireland (Bleasdale and Sheehy-Skeffington, 1994). The economics of grazing caused an increase in numbers of sheep and reindeer but management changes also encouraged soil erosion. In Britain and Ireland farmers were paid according to the number of animals they had. Sheep were shepherded less efficiently than they were before World War II, and hence they were allowed to congregate in particularly favoured areas to graze and rest. Winter feeding also allows more sheep to be kept on the hills. In Norway, subsidies for reindeer herders have been introduced in

TABLE 2.11.1 Grazing pressures in selected European countries[a]

Country	All animals[b] (animals ha^{-1})	Sheep and goats (animals ha^{-1})
UK	2.22	1.75[c]
Ireland	1.77	0.78[c]
Albania	1.45	1.12
Greece	1.17	1.11
Portugal	0.88	0.73
Macedonia	0.77	0.63[c]
Italy	0.68	0.42
Spain	0.67	0.53

[a]Data from *FAO Yearbook. Production 1999* (FAO, 2000). NB. The Netherlands also has high stocking densities (1.73 and 0.46) but the terrain is generally not conducive to erosion. Belgium/Luxembourg also has a high cattle stocking density (1.0) but again the terrain is generally not conducive to erosion.
[b]Horses, mules, asses, cattle, sheep and goats.
[c]No goats; ratios goats to sheep in other countries: Greece and Albania, 1:1.7; Portugal, 1:7.3; Italy, 1:8.0; Spain, 1:8.2.

the last few decades and the introduction of fencing to manage the reindeer confined them in certain localities so that maximum damage was inflicted upon the vegetation.

Runoff is greater from heavily grazed land (Evans, 1990b, 1998), especially from those areas particularly favoured by sheep (Meyles *et al.*, 2001), because the vegetation cover is very low or thin and soils are more compact. Furthermore, tracking and channeling of water downslope speed runoff into channels. Francis and Thornes (1990) do not state how shrub matorral in south-east Spain has become degraded, but grazing may well have played a part. Runoff and erosion are much greater under the poorer vegetation cover of the degraded matorral (35 % cover) than it is under undegraded matorral (49 % cover) and pine trees with a shrub understorey.

The hydrology of heavily grazed land is changed, therefore, and this is exacerbated if shrubs are grazed out, as in northern Norway, for example (Evans 1996a). Here, as in semi-arid parts of the world, gullying is taking place. Such a sequence of events could explain how some badlands in semi-arid Europe came about; once initiated it would be almost impossible to stop such soils gullying, especially if grazing intensities, as shown now by the tracked slopes, are kept high.

In the UK and Ireland there is good evidence, therefore, that animals have caused erosion and have increased runoff from the land and it may be no coincidence that grazing pressures are highest in these two countries (Table 2.11.1), both for all animals (pigs not included) and for sheep (and goats). Such high grazing pressures are also seen in the countries flanking the Mediterranean and could help explain the high sediment loads in rivers there.

2.11.6 AFFORESTATION AND DRAINAGE OF UNCULTIVATED LAND

In many parts of Europe, afforestation has been done carefully and there is little evidence of the deep ploughing that is done in Britain, or of the digging of drainage ditches, because neither of these have been needed. In Britain, however, the opening of furrows and ditches promoted rapid erosion. This was not recognised as a problem until the 1980s, even though the impacts on drinking water reservoirs were obvious almost from the first use of the large formerly redundant machines left over from World War II (Evans, 1996c). Even now, unless afforestation is done with care, erosion can occur and soil particles be transported into

streams. Rapid runoff from newly afforested ground can also cause downstream channel erosion (Johnson and Nisbet, in press).

Throughout Europe, however, wherever roads are opened up to plant and harvest trees, runoff and erosion of the tracks will have followed, to the detriment of stream channels. Harvesting of trees adjacent to river channels can also initiate bank collapse and erosion, which will continue for a number of years until the bank becomes stabilised again by vegetation.

In Britain, to improve the upland grazings, thousands of kilometres of ditches have been dug (Stewart and Lance, 1983). This has been going on for well over a century, but particularly after World War II until the mid-1980s when drainage was subsidised by the state. Even after a century, some of these drains can still be seen to be eroding and retreating up the hill. The ditches have greatly extended the drainage network. Such activity fortunately seems to have been rare elsewhere in the European uplands.

Planting eucalyptus woodland, sometimes in place of shrubs or olive groves, will have increased runoff and erosion greatly (Kosmas *et al.*, 1997) because soils below the trees are kept bare throughout the year, in addition to increasing the risk of fire (Grove and Rackham, 2001).

2.11.7 FIRE

Fires appear to play an important role in managing vegetation in two regions of Europe, the Mediterranean countries (Conacher and Sala, 1998; Grove and Rackham, 2001) and the British Isles (Anderson, 1997). Lightning-induced fires may have played a part in the evolution of some Mediterranean vegetation types and fires started by humans either maliciously or for the control of vegetation and for the improvement of grazing have been widespread in the Mediterranean basin for millennia. There is some evidence that the number of fires is increasing. Unless burns are very severe and consume all the vegetation cover, there is likely to be sufficient litter to protect the soil surface from severe erosion. Although the erosional impacts of burning can lead to severe erosion, as in an example in the USA (Moody and Martin, 2001), many burns in the Mediterranean region and elsewhere (Prosser and Williams, 1998) are not so severe, and their impacts may be very localised. Such a burn is described in Crete by Grove and Rackham (2001) that left a ground cover of 30 %, sufficient to protect the land from erosion. Also, the timing of rainfall in relation to the fire is critical. If a storm occurs immediately after the fire, erosion may be severe, but if it occurs after a period of time, the flush of plant growth using the nutrients supplied by the fire may be such that the ground surface is adequately protected from rainsplash and runoff.

Planting pine or eucalyptus trees, both susceptible to fire, may encourage more severe burns than under a ‘natural’ vegetation cover. This may be one of the reasons for the increasing incidence of fires in Portugal and Spain as 23 and 13 % of their forests, respectively, have been recently planted, a much higher proportion than other Mediterranean countries (FAO, 2001). In a fire-tolerant vegetation community fires may occur more often and the build-up of flammable material may never be sufficiently great to produce fires that are fierce and all-consuming (FAO, 2001; Grove and Rackham, 2001).

Fire is still frequently used in upland Britain as a management tool to encourage heather growth on moors. Such small burns provide a mosaic of vegetation types from low acid grasses to patches of old and young heather. Such a mosaic gives a wide range of food and cover for the grouse, a game bird reared to be shot. However, if not carried out skilfully burns can get out of hand. Burns caused accidentally or deliberately during prolonged periods of dry weather can be devastating (Imeson, 1971; Anderson, 1986, 1997; Alam and Harris, 1987) and trigger severe water and wind erosion. Many fires, however, as in the Mediterranean region, do not initiate erosion because they do not become hot enough to consume all the vegetation. Also, many moors are gently sloping and not vulnerable to water erosion. Until vegetation grows back, however, runoff may increase from such slopes.

2.11.8 ROADS, TRACKS, FOOTPATHS AND OTHER MECHANICAL DISTURBANCE OF THE GROUND

Roads and tracks opened up to afforest the land and harvest trees were mentioned above in connection with increasing runoff and initiating erosion. However, many roads have been built throughout Europe during the last 50 years, to improve movement between towns and to gain access to isolated communities, as in Norway, or to encourage tourism, as in many parts of the Mediterranean region. In upland Britain, many tracks have been driven across moors to allow easier access for managing sheep and for shooting game birds and deer. Often such roads are not, or were not at first, sealed. Rain rapidly runs off such surfaces, picking up sediment particles on the way. It has been shown that such forest and farm tracks (Froelich, 1995) can provide much of the sediment carried by streams. Roads and tracks opened up across hilly terrain often cut into hillsides and can destabilise slopes.

The development of tourism since the 1950s, especially hiking in the hills and mountains, has led to overuse and erosion of footpaths, not just in Europe but throughout the world (Liddle, 1997). Soils that are particularly vulnerable to footpath and vehicle tracking are peats, shallow screes, dune and heath sands and slopes in arctic and alpine localities. Gullies can form where soils are deep. Slopes steeper than 15° are especially vulnerable. Once footpaths are initiated, for instance on the Cairn Gorm, Scotland (Watson, 1985), they become wider over time and become more eroded unless usage for some reason declines (Lance *et al.*, 1989). Although such eroding tracks can constantly be repaired (Davies *et al.*, 1996), they are unsightly and channel water and sediment into streams in addition to destabilising slopes.

Skiing can also create bare soil and initiate erosion if the activity is prolonged into the spring period when melting snow exposes the underlying vegetation. Roads and tracks built to ski stations also expose the soil to the weather.

Possibly the greatest impact of machinery on uncultivated land has been the bulldozing of easily moved soils overlying soft, shattered or weathered rock in Mediterranean countries to allow the growing of vines, olives or almonds (see Chapter 2.12). Grove and Rackham (2001) consider the bulldozer to be the great driver of erosion in Mediterranean lands.

2.11.9 STORMS

Rare, large storms, such as those described in the Mediterranean region by Grove and Rackham, at least 300 mm in 36 h (Grove and Rackham, 2001), in northern England (100 mm up to 193 mm) by a number of researchers (Evans, 1996b) and other parts of Europe (and elsewhere; Starkel, 1976) can cause much damage to the land and river channels. Often, there is little that can be done to combat the impacts of very large storms. Such storms have often been exacerbated by human activities, for example forest clearance, overgrazing and construction of roads, tracks and drainageways. It is the lowering of thresholds by human activities that has made uncultivated land more vulnerable to the smaller and more frequently occurring storms. It is to protect land and property from these smaller storms that action should be taken. Such action will alleviate the impacts of large storms.

2.11.10 CONCLUSIONS

In this chapter, erosion of uncultivated land in Europe is described. Much of this land is in the hills and mountains where climate is more severe and land is used for grazing animals, forestry, water collection and storage and recreation. The modern (western) economy has a high demand for water and timber from these

uncultivated lands and tourism is an important part of the economy in many of these localities. Such economic demands can exacerbate, in addition to being affected by, soil erosion.

There is little good-quality information on the extent and severity of actual, not potential, erosion of uncultivated land in Europe and its impacts. There are four surveys of actual erosion of uncultivated land in Europe, one for Iceland (Arnalds *et al.*, 2001), one for Scotland (Grieve *et al.*, 1995) and two for England and Wales (Evans, 1990c; McHugh *et al.*, 2002). Evans (1996c) describes briefly the impacts of erosion of both uncultivated and cultivated land for England and Wales. There is an urgent need for a proper assessment of actual, not potential, erosion and its impacts in Europe.

Although much of the erosion in Iceland was undoubtedly initiated by human activities, only in the two surveys dealing with England and Wales is a distinction made between 'natural' and 'accelerated' erosion. Such a distinction is important if we are to tackle erosion and its impacts practically and within financial constraints.

We can do little to protect ourselves fully against rare, very large storms, or to combat erosion such as badland erosion, which now may be driven largely by natural processes although possibly initiated long ago by human activities. However, we can, if we wish, ameliorate the impacts of large storms and reduce greatly the impacts of smaller storms by how we treat the land.

It may be that erosion of uncultivated land has been studied more in Britain than elsewhere in Europe because the pressures exerted on soils and the landscape have been greater there. In other words, the problem of erosion of uncultivated land and its impacts has come to the fore first in Britain. Thus, when trees were planted after World War II to help reduce the reliance on imports, the excavation of drainage ditches was widespread. The UK has the largest extent, and proportion of its woodlands, in plantations in Europe (FAO, 2001). Similarly, the pressure to use the moors to produce more sheep led to the drainage of the peat. The 1947 Agriculture Act initiated headage payments on sheep and led to increasing numbers of them on the hills, and erosion followed. In Scotland, erosion has also accompanied the increasing number of red deer kept on the hills to be shot as game animals to bring income to the estates. Furthermore, the increasing standard of living after World War II led to the hills being trampled by walkers in large numbers from the 1960s onwards.

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2.12

Land Levelling

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2.12.1 INTRODUCTION

The preparation of a given area for a new type of crop or a different type of management has always involved modification of the local landscape through deforestation, terracing or bog and swamp reclamation. Nowadays, the great mechanical power of bulldozers (Figure 2.12.1a) is readily available to farmers and allows extensive land levelling. Hence, during the last 50 years, land levelling has gained an important role in European agriculture accompanied, however, by a considerable impact on the landscape.

Land levelling has been used in different situations: (a) reclamation of severely eroded areas, including badlands; (b) implementation of land use changes; (c) removal of unwanted vegetation (e.g. in abandoned lands); (d) removal/modification of field boundaries (e.g. the steps at field margins due to tillage erosion); (e) terracing (both in the making of new terraces and in the removal of old ones); (f) irrigation purposes; and (g) facilitating mechanisation in cultivation, etc.

Land levelling has been carried out whenever and wherever the profit (market trends and subsidies) was threatened by local morphology. Hence cereals, orchards, almond and olive groves and vineyards have all

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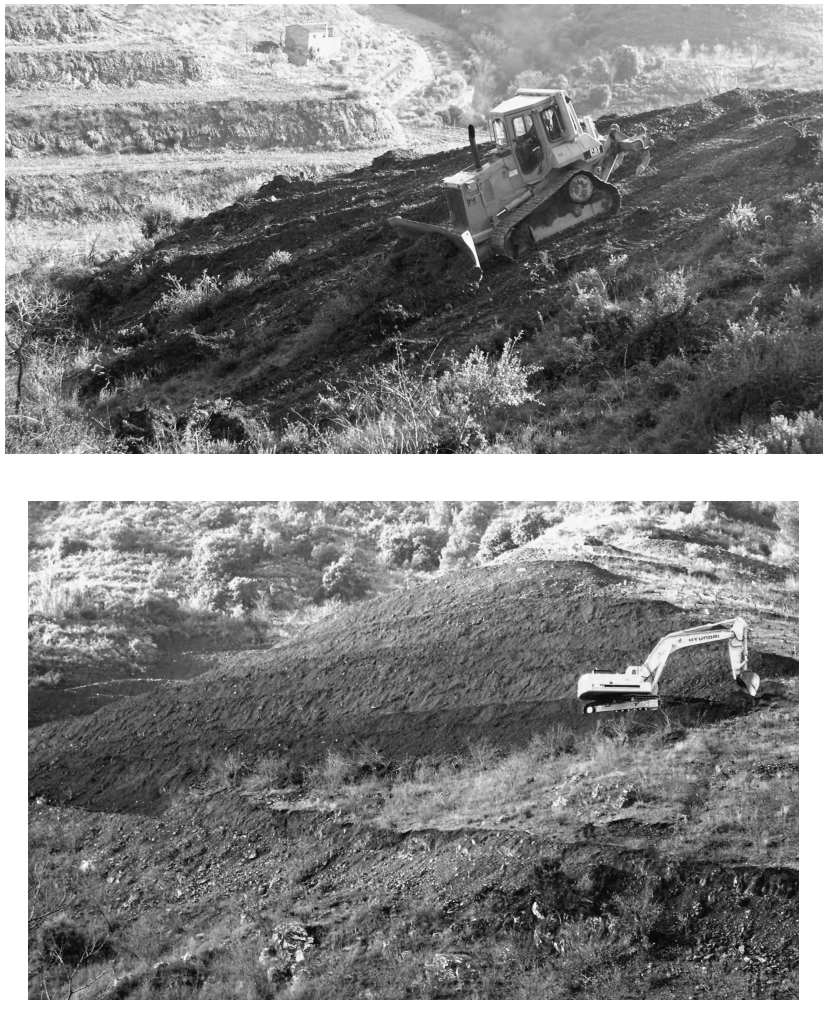

Figure 2.12.1 Bulldozers in action for (a) levelling and (b) building new terraces at Priorat (Catalunia, Spain)

called for levelling. Increasing the sizes of the once small fields that characterised most of the European agricultural landscape has usually been accompanied by levelling the structures of field borders.

Landscape is usually reshaped when the local morphology is sufficiently rough to impede cultural operations, including tillage, harvesting, de-weeding and fertilising. Bulldozers are also used to remove shrubs, vines, fruit trees or even structures such as small irrigation reservoirs. Scars made by landslides and ephemeral gullies are also removed using bulldozers. In some cases, levelling is associated with other operations, such as removal of impervious layers (to water and roots) and preparation of subsurface drainage systems. Land levelling by bulldozers is also used in 'traditional practices' as a substitute for manpower. For example, if new terraces are built, the new surfaces are designed using bulldozers. Sometimes, following the movement of large quantities of earth, the new slope appears subdivided into a few terraces with large (sometimes) sloping surfaces and high terrace rise (Figure 2.12.1b).

Land levelling spread considerably during the recent past and today is considered an almost common agricultural operation in many areas of Europe. As a consequence of this mental attitude, its effects went almost undetected and were underrated, despite their relevance. On the contrary, land levelling should be considered as a specific human-induced geomorphic process, which is often the most effective soil erosion process acting at field and hill-slope scale.

The characteristics and consequences of the implementation of land levelling vary with local constraints such as climate, morphology, geology and local legislation. Hence levelling has local characteristics also. It usually takes place within field borders and, therefore, it has a local impact. If substantial portions of small catchments are levelled, the hydrological and erosion response of the basin can be substantially modified, exporting local effects to another scale.

This chapter aims to present an overview of the many different ways in which land levelling is carried out and its interactions with other processes. Examples drawn from European countries will allow some generalisations and will give a clear perspective of the side-effects linked to such practice, thus proposing a European panorama, albeit to a considerable extent incomplete, of land levelling.

2.12.2 LAND LEVELLING IN EUROPE

The process of land levelling causes modifications of the landscape similar to those produced by prolonged tillage (see Chapter 2.9). The intensity of soil removal and soil deposition is obviously much greater, even when the material eroded/excavated and the amount of material deposited/accumulated remain inside the field border. Generally, there are two ways of approaching land levelling:

1. The soil is dispersed with disregard to soil characteristics so that pedogenic horizons and freshly fragmented parent material are mixed.
2. The more fertile top-soil (i.e. the soil top horizons, or the A–B horizons) is removed before levelling and later returned and spread back over the new reshaped surface.

The first approach was, until recently, the more common, whereas the second is either imposed (e.g. supported by central government, as in Norway) or found more convenient. Both are often accompanied by the installation of drainage systems for reducing the possible formation of perched water tables and mass movements. The reasons for land levelling are of various types and have followed different 'historical' paths in different countries, as will be evident from the following.

2.12.2.1 Norway

The first land levelling operation started in the 1950s using lightweight bulldozers. Land levelling peaked in the 1970s, especially in south-east Norway and partly around the Trondheimsfjord. Here, much of the agricultural area is situated on marine sediments. Natural erosion processes had resulted in a ravinized landscape, mainly used for grassland and pasture. In the 1970s, production systems turned into a continuous grain farming system. The change was promoted by political decisions and consequent subsidies which channelled grain production to lowland areas and animal production to inland and mountainous areas. Subsidies were introduced in 1972 for land levelling and former ravine landscapes used for pasture were turned into arable land usually ploughed in autumn. In the three counties around Oslo (Akershus, Østfold and Vestfold), the agricultural area given over to grain production increased from 29 to 80% in the period 1950–1980 as a result of the subsidy policy. Grain and milk prices were differentiated so that grain production was more profitable in the lowlands in south-east Norway and other districts with climatic conditions suitable for grain production.

Owing to the incentive of subsidies, land levelling really 'took of' and in 1973 an area of 3500 ha was levelled in 1 year. During the period 1950–85, about 35 000 ha were levelled, which is about 10 % of the total area used for grain production. In Akershus County, 25 % of the grain-producing area was levelled, with peaks of over 40% in some communities of the county.

This led to such serious and visible erosion problems (described in the following sections) that subsidies were withdrawn in 1985 and levelling was banned. New subsidies were also given to repair damage in levelled areas. It also prompted the start of soil erosion research in Norway, focusing especially on environmentally friendly production systems. Now, subsidies are given for reduced tillage, grassed waterways, buffer zones and sedimentation ponds.

When levelling started in the 1950s, the poor performance of lightweight bulldozers resulted in a cut-and-fill technique (Njøs and Slyngstad, 2001), where the old top soil down to the C- and R-horizons (0.6–0.9 m deep) ended up at the valley bottom underneath a new top layer. The top of the new soil profile was then a kind of C-horizon without organic matter, with negligible aggregate stability, low infiltrability, very low volumes of available water and air-filled porosity and high susceptibility to water erosion. When more powerful and heavier bulldozers came into use during the 1960s, it became possible to conserve the old topsoil, removing it before the levelling operation and putting it back later. However, the new topsoil is a mixture of subsoil and the old topsoil with reduced organic content, reduced aggregate stability and infiltration capacity and increased erodibility.

Levelling was necessary to transform steep land into land suitable for modern machinery. For example, safe operations of tractors and combines required land levelling of hills and ravines on slope gradients ranging between 15 and 30 %. In addition to levelling, in the same period, vegetation zones between agricultural land and creeks were reduced or removed by tillage. Meandering creeks were replaced by straight channels or piped.

Examples of bad land levelling and regulations, guidelines and suggestions for good land levelling are given in a single booklet (GEFO, 1988). However, recalling some of the indications given in the booklet is certainly instructive. For example, during levelling the following outcomes must be avoided:

1. To create a new topsoil with low infiltration capacity and/or too long levelled slopes (which sometimes have been found to exceed 400 m). These bring about intense surface runoff and concentrated flow in depressions and valley bottoms, enhancing rilling.
2. When inlets for surface runoff (installed for evacuating excess water through a pipe system) are lacking or not placed where water is actually concentrating (note that adjustments are often needed after the levelled area has settled), erosion problems around inlets/manholes are expected.
3. Instability and erosion of the slope (bank side slope) between the levelled area and the stream are enhanced when the local slope gradient is too steep. Lack of or scarce vegetation on this piece of slope and unchecked run-on can act as triggers.
4. If no stabilisation is made around the places where drainage pipes join the stream, high erosion rates can be triggered in its vicinity. This leads to breakage of pipes and catastrophic erosion in the fields.

Other procedures are instead recommendations. Some examples are as follows:

1. In order to avoid slopes being too long, the levelled terrain must be undulated with counter-slopes every 100 m to stop runoff and lead it to inlets (of an underground drainage system).
2. Inlets must be placed in such a way that settling of the levelled material does not make them protrude over the soil surface, interrupting the continuity of the water fluxes (0.5 m advised over-height on inlets). They should also be spaced at intervals of less than 100 m.
3. In order to stabilize the back side segment of the slope, the gradient must be less than 15 % and must be kept vegetated.
4. The outlets from the bottom pipes and the drainage system must be secured (e.g. armoured).

2.12.2.2 Germany

In north-east Germany (Ost Brandenburg), the major effect of land levelling was the loss of 'potholes' (Kalettka, 1996; Vahrson and Frielinghaus, 1998). Potholes are depressions (0.01–1 ha) formed by post-glacial melting of large blocks of ice over moraine deposit of the Pleistocene. They have an important role in hosting a large biodiversity, and they are the inlet to the main drainage system of the morainic deposits and outlet of large catchments.

Pothole frequency in this region varies between 0.6 and 40 potholes per km^2 (Kalettka *et al*., 2001). The pothole areas are located within the fields and all runoff and interflowing waters are retained in these depressions, maintaining a permanent or periodic water level, depending on the local hydrology. Unfortunately, potholes impede tillage activities and farm machinery traffic. Since the late 1950s, many have been filled and levelled and consequently up to 39 % of potholes in the region no longer exist (Kalettka *et al*., 2001). Kalettka (1996) reported the percentage loss of potholes also from other areas in Germany in the last 100 years, with up to 88 % in Schleswig-Holstein and 55 % in Alpen-Vorland. The large quantity of material needed for filling potholes was taken at the expense of the upper soil horizons of the surrounding areas causing a net loss of topsoil. The filling of potholes ended at the beginning of the 1990s after the unification of Germany.

2.12.2.3 Slovakia

In Slovakia, the collectivisation policy implemented mainly during the 1950s led to increased field dimensions and a different field pattern: the mosaic of originally small parcels vanished in favour of vast collectivised land units (Stankoviansky *et al*., 2000; Stankoviansky, 2003). The most intense terrain adjustments occurred with the levelling of former terraced plots. Also, the linear features (field borders) that had originated from different tillage intensities in contiguous fields were levelled.

2.12.2.4 Hungary

Land levelling was used in the areas where vineyards of high commercial value, producing most of the traditional vines, are situated. For example, in order to carry out mechanised cultivation of sloping vineyards, large-scale amelioration was carried out on the Badacsony hill (Lake Balaton) in the 1950s. One part of the present terraces was formed before World War II using individual methods. From the 1950s on, cooperative farming started a programme of activities (including land levelling) on the steeper slopes with a southern exposure. Usually these projects were sponsored by the State and were carried out with a low degree of accuracy. At the end of the amelioration activity, part of the territory was characterised by macro-terraces whereas the rest was cultivated downhill and protected by a grass cover. The more relevant consequences of these earth operations was that the A-horizon of the soil nearly entirely vanished, particularly in the case of macro-terraces, where the fertile soil horizon was buried under the embankment. Both methods provided protection for the upper soil layer against water erosion. Nevertheless, pipes and mass movements are still a real threat to terraced slopes.

2.12.2.5 Italy

Since the early 1950s, land levelling in Italy has historically been associated with two main goals. The first is to increase the arable lands by levelling of marginal lands, such as badlands. These operations were particularly intense in central and southern Italy, where Eocene and Pliocene sediments prevail, until the mid-1990s. The aim of these major movements of earth was to increase the surface available

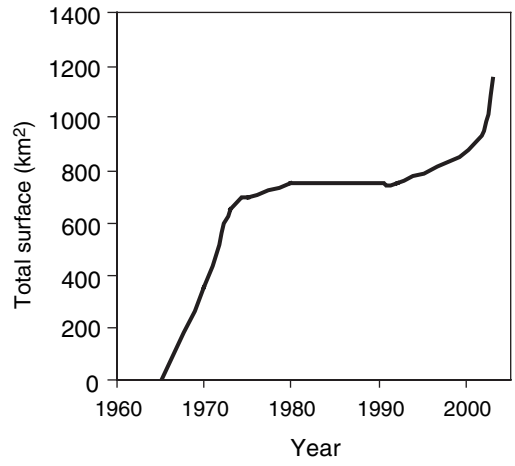

Figure 2.12.2 Increase in land levelled surfaces in Tuscany (Italy) since 1962 due to new vineyards

for cereal production (mainly durum wheat). In some areas such as in Tuscany (Guasparri, 1993; Phillips, 1998) and in Basilicata (Rendell, 1986; Phillips, 1998; Clarke and Rendell, 2000) most badlands were levelled (Figure 2.12.1) until the practice was finally banned in the 1990s (Regione Basilicata, 1994; Regione Toscana, 2001) because of their cultural and environmental value. Nevertheless, in some cases, owing to ineffective controls, land levelling still continues (Clarke and Rendell, 2000).

At present, land levelling always precedes the preparation of new vineyards, orchards and olive groves. The relationship is so strict that any variation of the total surface dedicated to these types of land use is a proxy for the area that is levelled. Recently, one of the authors estimated through GIS analysis that in Italy the area highly suitable for land levelling totals about 10 % of the area under permanent crops, corresponding to about 2522 km^2 (P Bazzoffi, unpublished work). The increment of vineyard acreage, shown in Figure 2.12.2, indicates the relevance of the process in Tuscany (similar trends are to be expected for every wine production zone in Italy). This pattern is correlated with the national and international wine market dynamics and also to EU Directives. Since 2001, the administration of Tuscany has been adopting the new European rules for the management of wine production (Regulation EU 1493/1999), allowing an increase in the total surface for high-quality wine to 1296 ha (about 10 % of the total surface assigned by the same act to the whole of Italy: 12 933 ha). Land levelling is only loosely controlled at present. Disciplining it lies at the 'Regione' administrative level. For example, the new forestry regulatory act of the Regione Toscana (2001), which disciplines these operations, permits levelling everywhere if the depth of the levelled layer is less than 50 cm. Hence to a large extent, levelling escapes any control. When the territory is classified as significant for hydro-geological protection of some area, a generic project and a report are required, certifying the hydro-geological compatibility of the levelling and surface reshaping. As no technical guidance of the type such as the Norwegian GEFO (1988) is available, procedures can be decided by incompetent persons.

Land levelling is usually carried out in summer or autumn. The general morphology produced by levelling operations, in the case of badlands, ranges from a uniform slope to a moderate rolling morphology, with a general reduction in the slope gradient. Smoothing and gradient reduction are more evident when the levelling is preliminary to new vineyards. Gradient reduction is usually obtained with cut-and-fill techniques with excavation of the upper part of the slope, scalping of local convexities, filling of main concavities and drainage

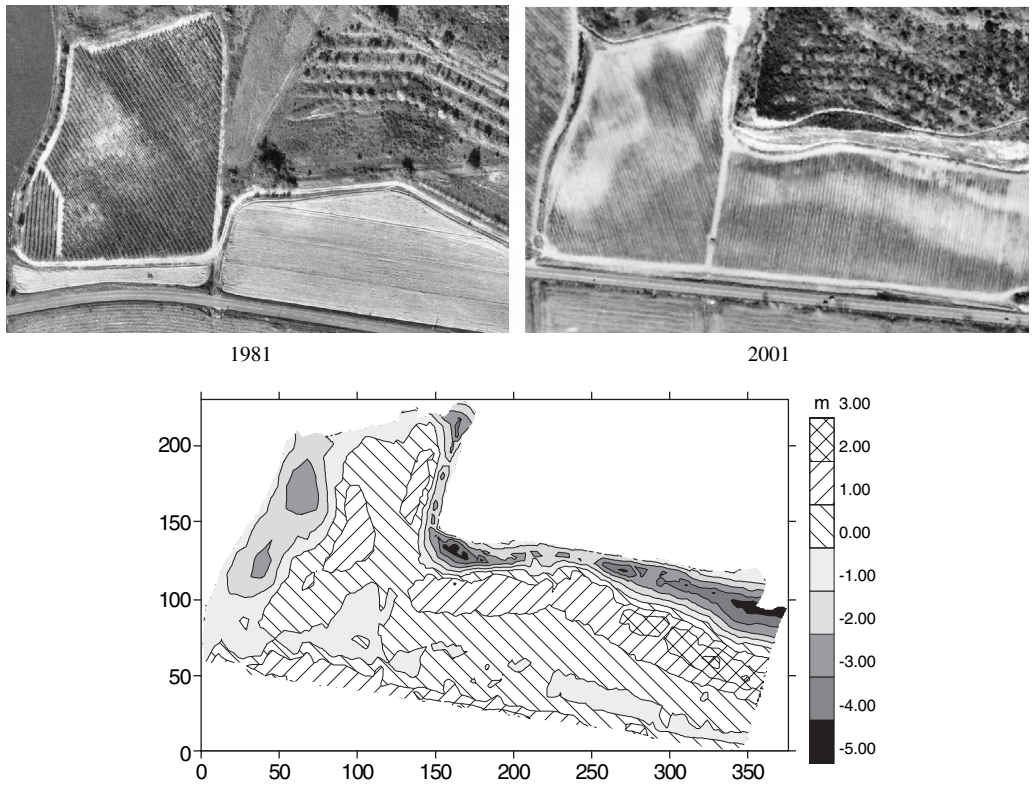

Figure 2.12.3 Land levelling for re-implantion of a vineyard (S. Gimignano, Italy): the differential digital terrain model (DTM) show from 3 to 4 m of erosion and accumulation in the various parts of the field (from Bazzoffi, 2002)

lines and accumulation downslope. The depth of excavation and accumulation ranges between 1 and 3 m in most cases, sometimes reaching values higher than 4 m (Figure 2.12.3; Bazzoffi, 2002). A sudden increase in slope gradient is produced at the upslope field border and a steep bank is formed at the downslope field border. Boulders and stones are removed before the final smoothing (see Figure 1.20.7). The loose remoulded material that covers the levelled area like a sort of uneven mattress is usually left uncompacted, waiting for natural compaction to act. Usually the new top layer is composed of crushed soft sedimentary rock mixed with original subsoil material. The top soil is rarely removed beforehand and then redistributed after the levelling operation: in this case the resulting top layer is made up of a mixture of the original pedogenic horizons. Deeper excavation occurs whenever deep drainage is designed to prevent landslides or reduce the possibility of local perched water table formation. In most cases the entire design of the land levelling operation is done on the bases of empirical knowledge with neither a specific geotechnical survey nor any design for a stable surface. Most of the procedures adopted are selected only considering constraints for future machinery operations.

The presence of pylons or telephone lines does not prevent land levelling (Figures 2.12.4 and 12.12.5a). Thus, poles become check points where, sometimes, even the original soil structure is preserved (Figure 2.12.4) or relicts of previous terracing system (Figure 2.12.5). These points can assist surveyors in the marking of the original surface profile and in the assessment of what has been definitely lost.

Figure 2.12.4 Evidence of soil scraping due to land levelling in a vineyard: the mounds around the poles indicate the amount of soil truncation (Tuscany, Italy)

2.12.2.6 Spain

The long history of land use and land-use changes that characterises Spain shows that people are one of the main agents producing the present landscapes (Lasanta, 1997). In Spain, the last period of expansion of the agricultural surface took place after the Spanish Civil War in the 1940s as a consequence of the post-War period of shortages. In marginal areas (e.g. southern Spain or mountainous lands), the traditional farming systems were basically of the extensive rainfed type including tree crops, mainly olive and almond groves. In steep slopes, the traditional plantations were placed on bench terraces. During the 1960s and 1970s, a progressive depopulation of the rural areas took place as a consequence of Spanish industrial development. Many terraced areas underwent progressive abandonment, with soil degradation and landscape deterioration (García-Ruiz, 1997; Faulkner *et al*., 2003). Later, in 1986, Spain joined the EU. Since then, the European Common Agricultural Policy (CAP) has been the main driving force behind agricultural development. The CAP measures decided 1992 and 1999 have led to an expansion of tree crops (mainly olives) in poorly developed rural areas. Thanks to subsidies, the total area used for olive tree plantations increased from 1.8×10^6 ha in 1989 to 2.3×10^6 ha in 1999. Subsidies were given for mechanisation, enhancing requests for land levelling, stone extraction and elimination of those subsurface soil layers that were limiting crop growth.

This obviously led to a spreading of land levelling, which in some places reached very high levels of intensity. Land levelling was consequently used for increasing field size, removing old terraces or creating new ones (Figures 2.12.1 and 2.12.5b), preparing new and modernising old plantations and vineyards. Also, introducing irrigation in fruit tree plantation (e.g. citrus) has called for land levelling.

Jiménez-Delgado *et al*., (2003) studied the area of L'alt Penedès – Anoia (Catalunia, Spain). They found that the amount of vinyards had increased to cover 30 % of the total surface in the region. They reported up to 5–8 m as the maximum excavation and accumulation depth in some areas. Here, too, land levelling was mainly employed to reduce slope gradient and facilitate machinery operations in the new vineyards.

As regulations exist regarding land-levelling procedures, decisions are taken by the project head engineer or directly by the field owner. As a consequence, the procedures applied are heterogeneous. Nevertheless, as

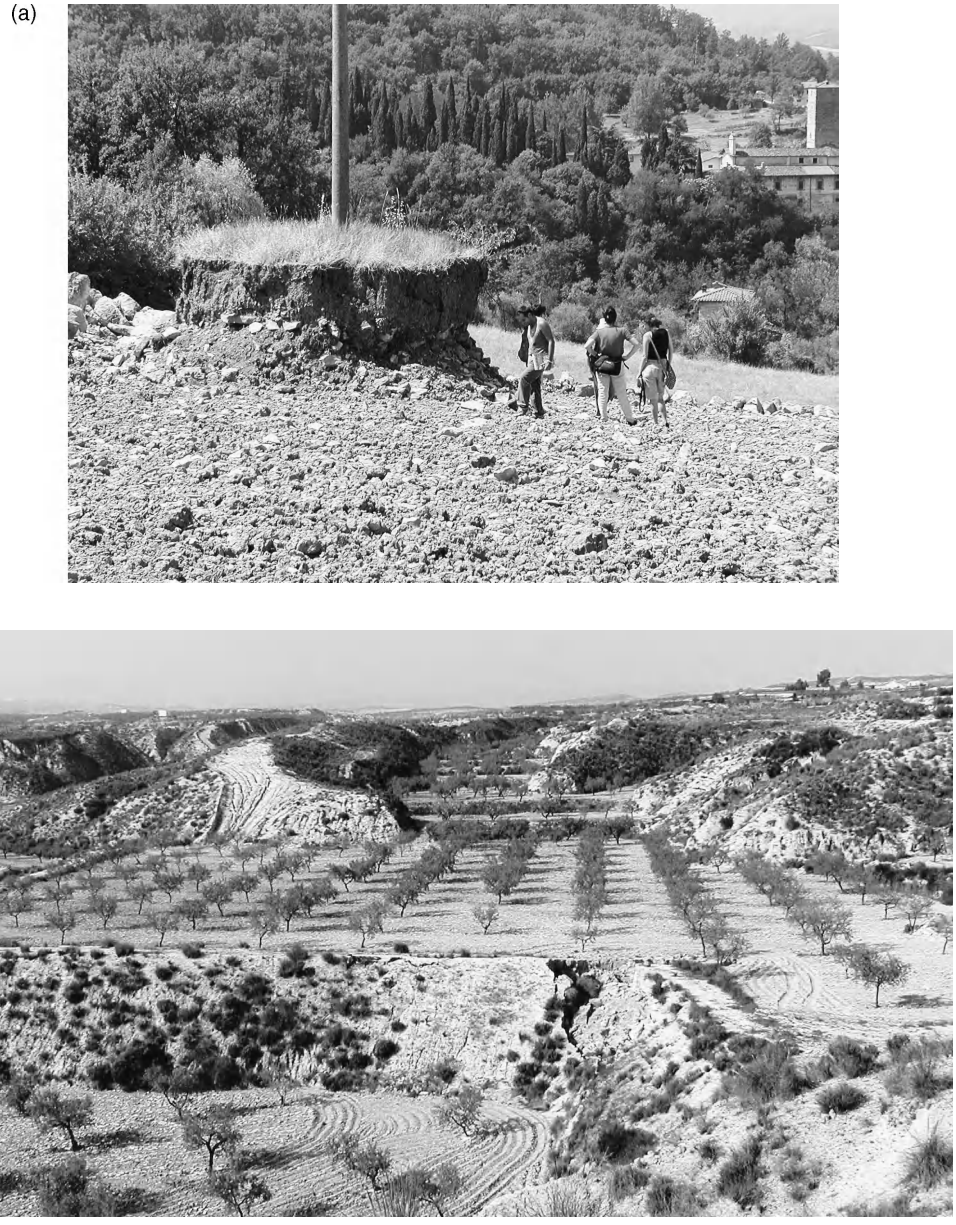

Figure 2.12.5 (a) Land levelling evidence (average excavation depth: 3 m) in the upper Arno Valley (Tuscany, Italy) where an old terracing system (mean wall height: 2.5 m) was levelled (10 ha) for a new vineyard in 2002. (b) New terracing system in Murcia (Spain): a gully connects the last two terraces

a reference, a detailed description of the methods and techniques for land levelling and the proper management of the original topsoil ('Capaceo') was given by the Spanish Ministry of Agriculture (Cano-Muñoz and Vazquez-Guzman, 1997). When the land levelling operations affect non-cultivated areas with natural vegetation (forest, shrubs or permanent grasslands), a detailed report of the entire project is required,

which is then reviewed to assess its environmental impact. The assessment process implies the obligation for the land levelling project to include all the necessary/required measures to ensure the conservation of the topsoil.

2.12.3 ON- AND OFF-SITE EFFECTS

2.12.3.1 Landscape and Morphology

Land levelling is a source of permanent changes in the local landscape. As both soil and hill-slope are affected by land levelling, results are the loss of a series of functions as a counterpart for a slope more suitable for agriculture. The new landscape is usually less varied than the old one (e.g. vineyards and forest where once there was a more complex landscape pattern based on mixed farming with cereals and fodder in between olive grove and orchard, alternating with small vineyard plots). Hence we usually small different niches are lost, substituted by a large cultural unit, with a net loss of biodiversity. Entire landscape types, such as the biancane badlands in Italy, risk disappearance.

2.12.3.2 Soil Profile

The mixing of the top horizon with the other soil horizons generally causes deterioration of the physical, chemical and biological characteristics of the resulting soil. This is because the subsurface horizons have a structure which is often less porous, richer in clay and with lower aggregate stability, especially with respect to the action of rain water (increased dispersion of particles, hence more intense sealing), than the topmost horizon (Øygarden *et al*., 1997; Phillips, 1998; Robinson and Phillips, 2001; Nacci *et al*., 2002; Torri *et al*., 2002; Lundekvam *et al*., 2003). Occasionally, the reverse way occur.

The original soil structure is always lost in any type of land levelling. The upper soil horizons are either redistributed over the new slope surface or are irremediably mixed with crushed subsurface material. Hence deterioration of soil hydrological and erosion characteristics directly follow land levelling. Also, soil fertility is reduced, as can easily be observed in the field: vegetation is often healthier in those spots where soil material was accumulated.

Obviously, as the Norwegian experience teaches, redistributing the top-soil after levelling alleviates problems. Njøs and Hove (1984) demonstrated the importance of conserving the organic matter at the soil surface. Simply covering the surface with straw or incorporating manure or sewage sludge can significantly reduce erosion. They also stated that levelling would have been a safe investment if some form of rotation had continued. In Italy, the reclaimed (levelled) badlands are exposed to crusting because of the negligible aggregate stability and the high crusting potential (Robinson and Phillips, 2001; Torri *et al*., 2002). Phillips (1998) indicated a threshold of 2 % of organic matter and 15 % of exchangeable sodium for preventing intense erosion in reclaimed badlands on pliocenic sediments in Italy.

2.12.3.3 Hill-slope Hydrology

The modifications of the soil mentioned above affect the partition of rainfall into overland flow and subsurface flow (Øygarden *et al*., 1997), thus modifying the hydrological response. Note that soil removal and remixing are not accompanied by any consolidation and compaction of the remoulded layers. This causes the presence of large pores, which favour intense subsurface fluxes and erosion. These large pores will disappear during the reconsolidation of the slope.

When levelling is coupled with measures such as drainage, there can be a stabilisation of the slope with beneficial effects. Absence of stabilisation measures caused an increased hazard of mass movement (Torri *et al*., 2002).

2.12.3.4 Soil Erosion by Water and Slope Stability

If not accompanied by preventive measures, the modification of slope morphology, removal of convexities and filling of concavities can destabilise the equilibrium along the slope (Ballerini *et al*., 1991; Torri *et al*., 2002). Examples are the substitution of a rill network with a single gully, obviously ephemeral, or local destabilisation of the slope with increased frequency of shallow mass movements (Figure 2.12.6). The presence of intense water fluxes in the very large macroporosity already discussed can cause intense tunnel erosion, which is often experienced as a sudden lowering of the soil surface when tractors pass over tunnel roofs. Piping is often so intense that this often neglected erosion process is probably the most important cause of soil loss in levelled surfaces.

It must be stressed that these types of 'catastrophic' events cannot be tolerated because the workability and trafficability of the field/slope will be strongly reduced. Hence bulldozers will be further used for reshaping the eroded spots or soil material will be imported from elsewhere to refill holes, scars and depressions. Gullies and mass movements can cause the exposure of the roots of recently transplanted

Figure 2.12.6 Large landslide in the lower part of a new (and freshly levelled) vineyard in the Chianti area (Tuscany, Italy). The landslide crown is approximately 40 m wide and the main scarp is 2 m high

plants (such as vines). Sometimes, plants can be completely uprooted. If the damage is too great then replanting is also needed, which will prolong the period of surface instability of the slope. All levelled portions in a hilly landscape are generally fairly erodible and medium to high magnitude storms can provoke catastrophic erosion. Lundekvam (2002) estimated that soil erodibility may increase from 3- to 13-fold after levelling, depending on the quality of the levelling operation. The long period during which a levelled slope remains bare makes it almost inevitable that one or more erosive events will take place when the slope is more erodible. This is well exemplified for Italy (Chapter 1.20). When extreme events occur, the results can be incredible. Jiménez-Delgado *et al*., (2003) reported that the area of L'alt Penedès – Anoia (Catalunia, Spain) experienced an extreme single erosive storm (105-year return period) in 2001. It caused massive soil erosion (up 400 Mg ha^{-1}) and considerable loss of nutrients (Ramos and Martínez-Casasnova, 2004). These large values are a further reason making any concept of tolerable soil loss irrelevant.

2.12.3.5 Off-site Consequences

Erosion introduces off-site consequences of land levelling and associated control measures. On the basis of the above consequences of land levelling, one can expect some changes in the amount and way slopes export water and soil.

During the first year, a large export of soil material is to be expected with the creation of large gullies. This can cause off-site problems depending on the utilisation of the water downstream of the source slope. This type of problem should then decline with time as the levelled slope reconsolidates and vegetation recovers.

The modification of the surface network of the concentration lines of overland flow is permanent. Hence slope–channel connectivity must also change. The same is true for subsurface flow, especially if it is intercepted by a new system of underground drains. In particular, this can change the speed at which subsurface water will join the channel network, with obvious consequences on peak time and rate. In some cases this will not dramatically affect the situation, in others the catchment threshold for floods can be seriously lowered. When the lowering of the flood threshold is coupled with a bare soil surface, as is the case during the first year after levelling, then unusual and unexpected muddy flow can be triggered, with obvious consequences downstream. Hence increased rate of silting of creeks, rivers and lakes and of eutrophication of water bodies can be expected.

The risks described above can increase dramatically if land levelling is applied to a considerable portion of a basin.

2.12.3.6 Evaluating the Impact of Land Levelling on Risk Assessment

The practices of land levelling described above modify the slope/catchment response to precipitations and erosion to an extent that can vary from beneficial to highly hazardous. Evaluations of the possible impact of levelling can be assessed qualitatively at least. The use of the diagrams in Figure 2.12.7 may help. A first criterion of assessment is based on the type of techniques and design of land levelling adopted (Figure 2.12.7a). The second criterion is based on the on-site effects after the operations of earth displacement (Figure 2.12.7b). For example, a tolerable instance of land levelling would be one that minimises change to the slope gradient. Usually land levelling is carried out in such a way as to fall between the two extremes, often closer to the worst one. The two extreme classes are defined by trying to synthesise the experience that has been discussed earlier and considering the known interaction between slope hydrology, soil erosion and slope stability.

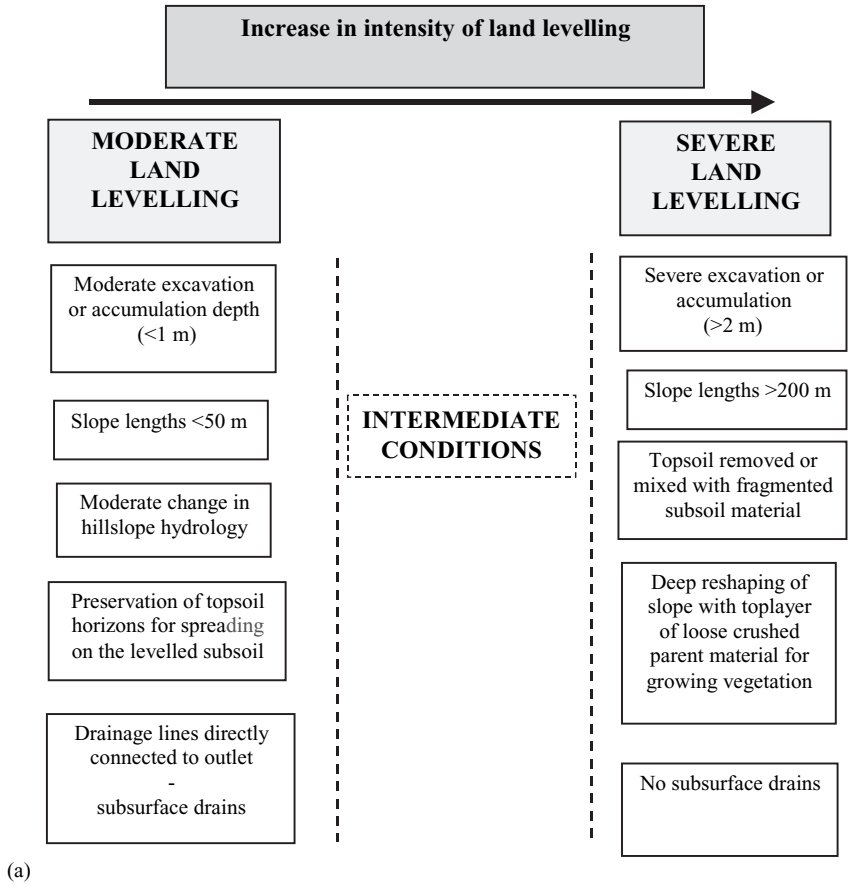
Increase in intensity of land levelling
MODERATE LAND LEVELLING
SEVERE LAND LEVELLING
Moderate excavation or accumulation depth (<1 m)
Slope lengths <50 m
Moderate change in hillslope hydrology
Preservation of topsoil horizons for spreading on the levelled subsoil
Drainage lines directly connected to outlet - subsurface drains
INTERMEDIATE CONDITIONS
Severe excavation or accumulation (>2 m)
Slope lengths >200 m
Topsoil removed or mixed with fragmented subsoil material
Deep reshaping of slope with toplayer of loose crushed parent material for growing vegetation
No subsurface drains
(a)

Figure 2.12.7 Qualitative assessment of the impact of land levelling on associated risks

2.12.4 CONCLUSIONS

Land levelling is an important and underrated process of landscape modification. It has usually been neglected until now (Norway representing the exception) because it is considered a necessary operation for improving agriculture and landscape features. Land levelling is used in every European country whenever the surface is rough. It is primarily motivated by a set of factors, such as the price dynamics of the agricultural products, and by European and national policies and subsidies.

In addition to satisfying its primary goal (easy and safe to manage fields), land levelling causes permanent losses such as the original pedostructure and landscape. Increased risk of concentrated erosion always accompanies land levelling owing to the increase in erodibility of the new top-soil material. Landslide risk increases when no specific measures for improving slope stability (e.g. deep drainage or retaining structures) are implemented. Moreover, piping, an often neglected erosion process, produces effects which are at least as important as rill and gully erosion. Obviously, once levelling has disrupted the soil, the concept of tolerable soil loss is invalidated.

The new reshaped and levelled surface poses considerable problems for off-site consequences linked to sediment and nutrient delivery outside fields, particularly during the first 2 years after levelling.

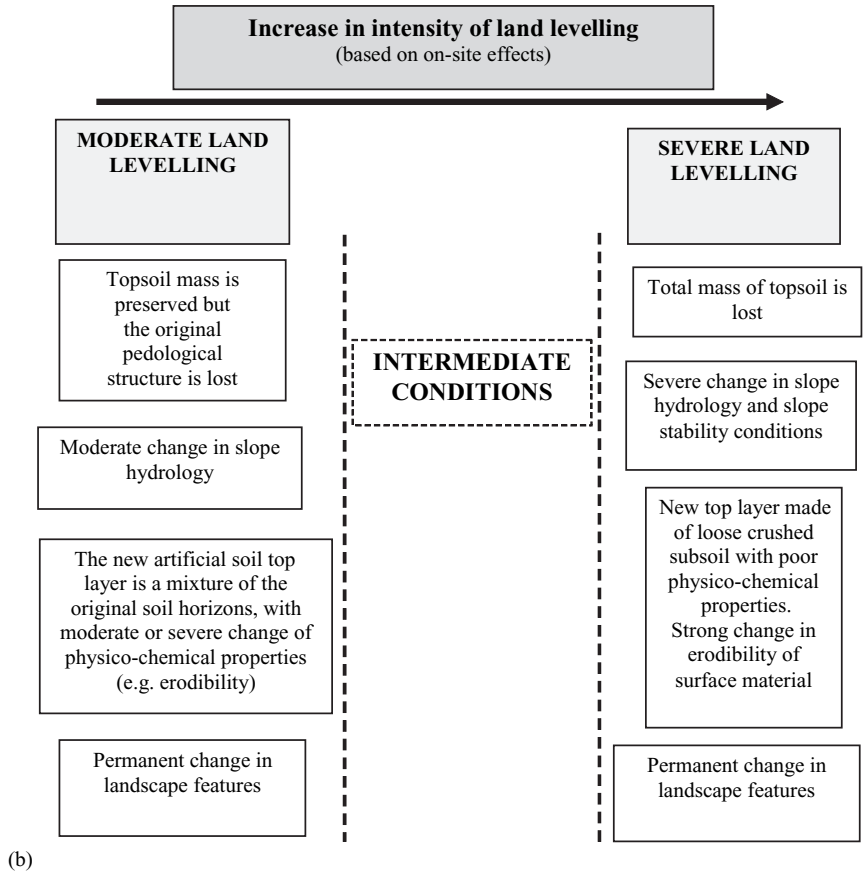
Increase in intensity of land levelling
(based on on-site effects)
MODERATE LAND LEVELLING
SEVERE LAND LEVELLING
Topsoil mass is preserved but the original pedological structure is lost
Total mass of topsoil is lost
INTERMEDIATE CONDITIONS
Moderate change in slope hydrology
Severe change in slope hydrology and slope stability conditions
The new artificial soil top layer is a mixture of the original soil horizons, with moderate or severe change of physico-chemical properties (e.g. erodibility)
New top layer made of loose crushed subsoil with poor physico-chemical properties. Strong change in erodibility of surface material
Permanent change in landscape features
Permanent change in landscape features
(b)

Figure 2.12.7 (*Continued*)

A better understanding of the feedbacks associated with levelling needs further study and data. The information on which this chapter is based shows that the risks associated with this practice are fairly high and that the practice itself is common. In this light, it becomes evident that land levelling is unquestionably a European problem. Research to expand the knowledge on this topic is needed together with more complete data sets as a platform for introducing sound regulation.

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2.13

Pan-European Soil Erosion Assessment and Maps

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2.13.1 INTRODUCTION

Soil erosion is a natural process, occurring over geological time, and most concerns about erosion are related to accelerated erosion, where the natural rate has been significantly increased by human activity. Soil erosion poses severe limitations to sustainable agricultural land use, as it reduces on-farm soil productivity and causes the accumulation of sediments and agro-chemicals in waterways. In Europe, soil erosion is caused mainly by water and, to a lesser extent, by wind. Rill and inter-rill erosion affects the largest area, but evidence of rills and inter-rills can easily be hidden by normal tillage. Gully erosion and landslides, on the other hand, often scar entire landscapes but are relatively localised. Soil losses due to water erosion are high in southern Europe and moderate in northern Europe.

Factors such as climate, topography, specific soil characteristics, land cover and management have important effects on both runoff and the process of soil erosion. Depending on these factors, average human-induced soil erosion rates, due to rill and inter-rill erosion, are typically between 1 and 26 t ha $^{-1}$yr $^{-1}$ for Mediterranean arable land (Tropeano, 1983; Kosmas *et al.*, 1997) and between 0.5 and 7.8 t ha $^{-1}$yr $^{-1}$ for arable fields in northern Europe (Bollinne, 1982; Kwaad, 1994; Chambers and Garwood, 2000). On the other hand, soil formation is a slow process involving the breakdown of rock into small particles and the accumulation of organic matter. Compared with slow soil formation rates, soil can be regarded as a nonrenewable resource. These are compelling reasons for assessing and monitoring soil erosion and improving the way soils are managed.

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Both national and international agencies need objective spatial information at low resolutions to compare levels of environmental risk and focus policies on environmentally sensitive areas. The dynamic relationship between human activities and the environment requires that environmental processes such as erosion be quantified to monitor and evaluate the impact of agriculture and land-use policies. Policy-makers need to know the area affected by soil erosion and an estimate of the magnitude at a regional scale in order to formulate suitable remediation measures and mitigation strategies (Gobin *et al.*, 2003, 2004).

This chapter deals with methods to present and assess the extent of soil erosion by water at a pan-European scale. Seven different methods and maps are compared for carrying out pan-European soil erosion assessment. The GLASOD and HOT SPOTS maps are based on distributed data and expert judgement. The RIVM/IMAGE, CORINE, ESB/USLE and INRA methods are based on ranked factors and the empirical (R)USLE model. The PESERA map is the result of a process modelling approach to assess regional soil erosion risk.

2.13.2 REGIONAL ASSESSMENT METHODS

Regional assessment methods of soil erosion can be distinguished as distributed point data involving expert judgement, indicator or factorial approaches and process modelling. All of these methods require calibration and validation, although the type of validation needed is different for each category. There are also differences in the extent to which the assessment methods identify past erosion and an already degraded soil resource, as opposed to risks of future erosion, under either present climate or land use, or under scenarios of global change.

2.13.2.1 Methods Using Distributed Point Data

An important form of erosion assessment is from direct field observations of erosion features and soil profile truncation. Erosion features consist of rills and gullies, some of these ephemeral, and of associated deposition in swales and small valleys. Soil profiles may show local loss of upper horizons, or burial by deposition from upslope.

Measurements of soil erosion represent a second important source of information. However, there are only a limited number of datasets available from experimental erosion plot sites and catchments in Europe. Most of these datasets cover periods of short duration, i.e. up to 3 years.

Data may also be collated from remote sensing surveys of erosion features, using aerial photographs or other earth observation techniques. Satellite images with resolutions of 25–30 m, e.g. LANDSAT, can be used to detect severe and distinct forms of land degradation and soil erosion. Very high-resolution imagery such as IKONOS permits the detection of rills and smaller gullies. Aerial photographs at different temporal and/or spatial scales may serve the same purpose.

Data in the form of measurements, field observations or remote sensing surveys may also be collected from regional experts in soils or soil erosion. Questionnaire surveys may be administered to the scientific community and national soil experts to arrive at a regional assessment based solely on expert judgement.

All these distributed data methods require validation to standardise differences in the intensity of study of different areas and in the clarity of suitable features on different soil types. There are also differences in methods and traditions between scientists in different areas of Europe. On their own, these methods cannot provide a complete picture except for small sample areas, and require the use of other methods to interpolate between areas.

The main advantage of distributed observations or measurements of erosion is that data are unambiguous where they exist, and give a good indication of the current state of degradation of the soil resource. The main

disadvantage of these methods is that they provide little or no information about when erosion occurred, unless there are supporting data on this point.

2.13.2.2 Indicator or Factorial Approaches

Since many of the processes and factors that influence the rate of erosion are well known, it is possible to rank individual factors for susceptibility to erosion, providing a series of erosion indicators. For example, soil indicators reflect the tendency for crusting, the experimental erodibility of soil particles and aggregate stability. Similar rank indicators can be developed for parent materials. Climatic indices are based on the frequency of high-intensity precipitation and on the extent of aridity or rainfall seasonality. Topographic indicators incorporate slope gradient and contributing area, a value corresponding to the catchment area upslope of a point. Vegetation indicators consist of land cover, vegetation type and roughness. Land management indicators comprise tillage operations, crop selection, spacing, orientation with the slope and other field practices. Clearly, a high susceptibility for all factors indicates a high erosion risk and a low susceptibility for all factors indicates a low erosion risk.

Individual indicators can be mapped separately, but it is more problematic to combine the factors into a single scale, by adding or multiplying suitably weighted indicators for each individual factor. There are difficulties both about how to select and justify the individual weightings and about the assumed linearity and statistical independence of the separate factors. The method should therefore be most effective for identifying the extremes of high and low erosion, but less satisfactory in identifying the gradation between the extremes. Despite these theoretical limitations, factor or indicator mapping has the considerable advantage that it can be widely applied using GIS datasets. For example, raster GIS coverages of topography, soils, land use and climate are readily available for Europe.

There is a wide spectrum of possibilities to use some or all of the factors of the Universal Soil Loss Equation (USLE), where soil loss is a simple multiplication of five factors: soil erodibility, rainfall erosivity, slope length factor, vegetation cover factor and crop management practice. However, all of the approaches remain an imperfect implementation of the USLE, partly owing to the historical lack of systematic data in Europe.

2.13.2.3 Process Modelling

A third approach towards Europe-wide soil erosion assessment is the application of a process model. A process model consists of components with an explicit physical basis. Current thinking on soil erosion modelling recognises the importance of runoff forecasting as a critical control on erosion loss. Runoff models are based on a runoff threshold or infiltration equation approach, and vary in complexity from the RDI model (Kirkby *et al.*, 2000) to the USDA WEPP model (Nearing *et al.*, 1989). For application at the regional scale, most erosion models are severely limited by their high data demand and, in many cases, by a focus on individual events rather than long-term cumulative impacts.

Process models have the potential to respond explicitly and rationally to changes in climate or land use, and so have great promise for developing scenarios of change and what-if analyses of policy options. Set against this advantage, process models generally make no assessment of degradation up to the present time, and can only incorporate the impact of past erosion where this is recorded in other data, such as soil databases. Models also generally simplify the set of processes operating, so that they may not be appropriate under particular local circumstances. Although, *prima facie*, the modelling approach appears to be the most generally applicable, there are major problems of validation, and in particular in relating coarse-scale forecasts to available erosion rate data, much of which is for small erosion plots.

A process model that is suitable for regional soil erosion assessment should represent the state of the art in current understanding of soil erosion processes, combine sufficient simplicity for application at a European

scale and have the potential for downscaling to field scales for purposes of explicit validation. PESERA offers such a methodology (Gobin *et al.*, 2004).

2.13.3 RESULTS OF PAN-EUROPEAN ASSESSMENT METHODS

2.13.3.1 The GLASOD Approach

The main objective of the worldwide GLASOD (Global Assessment of the Current Status of Human-Induced Soil Degradation) Project was to strengthen the awareness of decision makers on the risks resulting from inappropriate land and soil management (Oldeman *et al.*, 1991). On the basis of incomplete existing knowledge, a scientifically credible global assessment of the geographical distribution and the severity of human-induced soil degradation was made within a very short time frame. The task was subcontracted to correlators in 21 regions to prepare, in close cooperation with 300 national soil scientists, regional soil degradation status maps. All collaborators were provided with guidelines and a base map with loosely defined physiographic units. The assessment consisted of an expert judgement according to the general guidelines of degradation status (type, extent, degree, rate and cause) for individual polygons on a national/sub-national level. The regional maps were compiled, correlated and digitised to provide the GLASOD world map of soil degradation.

The European part of the GLASOD map has been updated on the basis of questionnaires that were sent to scientific teams in each European country for comments and additions (van Lynden, 1995; Figure 2.13.1). Not all countries completed and returned the questionnaires and the degree of detail of the information received varies greatly. It must also be noted that the scale of the maps (1:10 000 000) limits the detail that can be

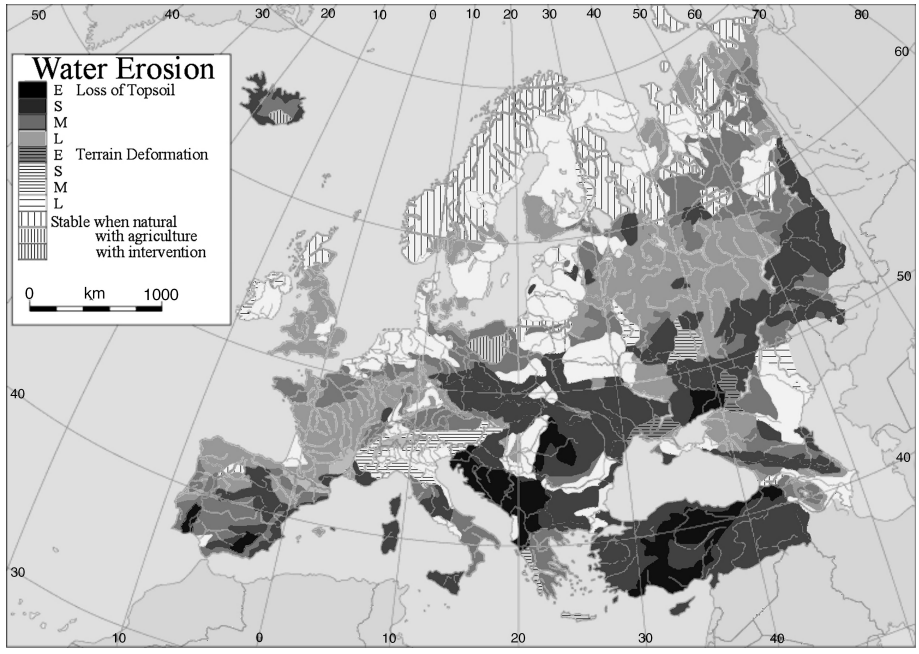

Figure 2.13.1 Water erosion of soils in Europe according to the GLASOD approach (modified from Van Lynden, 1995, © Council of Europe, reproduced with permission)

shown, providing a minimum resolution of approximately 10 km. The representation of the map items causes a visual exaggeration: each polygon which is not 100 % stable shows a degradation colour, even if only 1–5 % of the area is actually affected. The GLASOD map identifies areas with a subjectively similar severity of erosion, irrespective of the conditions, which produced the erosion, and irrespective of its wider impacts.

Despite the limited aims of the project and the subjective approach, GLASOD is the only method that has been applied at a worldwide scale (Oldeman *et al*., 1991). Therefore, the GLASOD map and complementary statistics have been used and cited in numerous scientific journals and policy documents of the World Resources Institute, the International Food Policy Research Institute, the Food and Agriculture Organization of the United Nations, the United Nations Environment Programme and many others. However, the dependence on a set of expert judgements results in very little control or objectivity in comparing the standards applied by different experts for different areas.

2.13.3.2 The HOT-SPOTS Approach

An analysis and mapping of soil problem areas (hot-spots) in Europe was published in the EEA–UNEP joint message on soil (EEA and UNEP, 2000). The purpose of the study was to support the joint message on the need for a pan-European policy on soil, identifying 'hot-spots' of degradation in Europe and examining environmental impacts leading to change and particularly degradation of soil function. The hot-spots map for soil erosion aims to present a kind of 'spatial indicator' that would permit the identification of priorities of intervention and the visualisation of data gaps.

The temporal and spatial patchiness of soil erosion and the uneven density and quality of local measurements make a simple mapping of hot-spots futile. Soil erosion in the hot-spots approach is therefore indicated at a hierarchy of scales (Figure 2.13.2). Three zones are identified for which the nature of erosion is generally similar: Eastern Europe, the Loess Belt and southern Europe, which primarily represent different land-use history, parent materials and climate, respectively. Hot-spot areas are then highlighted within each zone on the basis of two earlier maps (de Ploey, 1989; Favis-Mortlock and Boardman, 1999). The intention is to identify areas of current erosion risk, under present land use and climate. Hot-spot locations, mostly within the hot-spot areas, are sites with available measurements of erosion rates, taken from erosion plots, fields and small catchments.

Worth noting is that the underlying map indicating hot-spots is an extension of the soil erosion risk map of Western Europe by de Ploey (1989). Various experts were consulted to identify areas where, according to their judgement, erosion processes are important. A limitation of the de Ploey map is that there are no clear definitions of the criteria according to which areas were delineated and hence there is no assurance of objectivity.

2.13.3.3 The IMAGE/RIVM Approach

A baseline assessment of water erosion was prepared for 1990, as part of a major report on strategies for the European Environment (RIVM, 1992). The assessment of current risk was combined with climate and economic projections within the framework of the IMAGE 2 model to generate scenario projections for 2010 and 2050. IMAGE 2 is an integrated model designed to simulate the dynamics of the global society–biosphere–climate system (Alcamo, 1994).

Water erosion represents a module of the IMAGE model adapted from the water erosion model of Batjes (1996) on a $\frac{1}{2}^{\circ} \times \frac{1}{2}^{\circ}$ (approximately 50 km) grid (Figure 2.13.3). The water erosion impact module generates a water erosion risk index based on three main parameters: terrain erodibility, rainfall erosivity and land use. Terrain erodibility is based on soil type and landform. Landform is classified into types by using the difference between minimum and maximum altitudes for each grid cell. Soil type is derived from the FAO Soil Map of

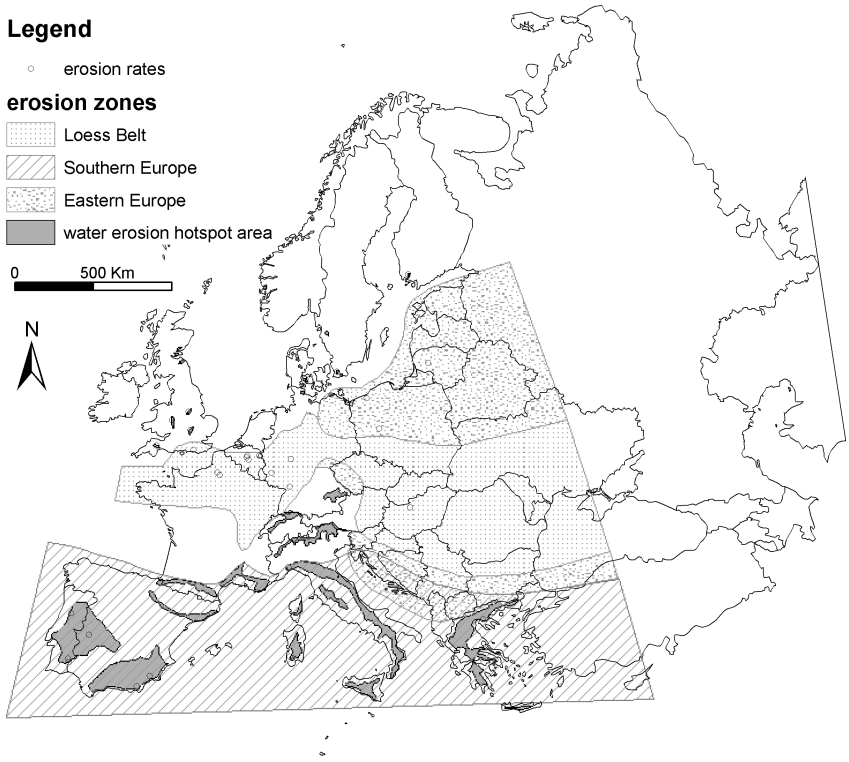

Figure 2.13.2 The soil erosion hots-pots map (modified after EEA and UNEP, 2000. Assessment and reporting on soil erosion. Background and workshop report. Technical report nr. 94. European Environment Agency, reproduced with permission)

the World and is assessed on the basis of soil depth, soil texture and bulk density. Rainfall erosivity is derived from the monthly maximum rainfall per rain-day. Data on precipitation and number of wet days are derived from the IIASA climate database. The potential erosion risk derived from terrain erodibility and rainfall erosivity is converted to actual erosion risk by a land cover factor, representing the degree of protection afforded by various land covers. Natural vegetation with a closed canopy, e.g. forests, is assumed to provide optimal protection, i.e. no risk. Natural vegetation with a more open structure, e.g. shrubs, is assumed to provide sub-optimal protection, i.e. low risk. Highest risks are assigned to arable land, linked to a crop protection factor. Land cover maps for the IMAGE model are derived from several sources including Olson's land cover database and statistical information from FAO.

The IMAGE/RIVM approach has the advantages of making explicit scenario projections and combining physical and economic elements within a single framework, but the 50-km resolution renders it difficult to interpret and validate at sub-national scales.

2.13.3.4 The CORINE Approach

The CORINE programme was established in 1985 to help incorporate an environmental dimension into Community Policies, to ensure optimum use of resources to obtain environmental information and to develop the methodological base needed to obtain environmental data, which are comparable at Community level. For one of the priority topics, soil erosion, a new methodology was developed, which provides a factor-based

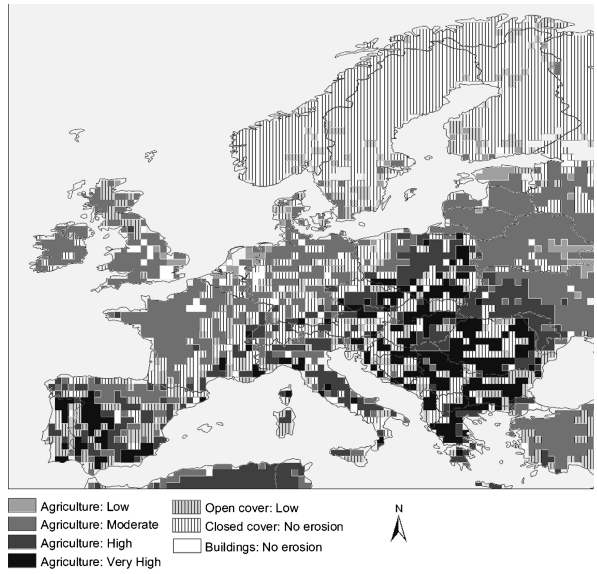

Figure 2.13.3 Water erosion vulnerability for 1990 (modified from RIVM Report 481505001. The Environment in Europe: A Global Perspective. Copyright 1992 RIVM, reproduced by permission of RIVM)

assessment of risk. The CORINE soil erosion methodology (CORINE, 1992; Briggs and Giordano, 1995) was based on a considerable simplification of the USLE.

The CORINE soil erosion risk maps are the result of an overlay analysis of factorial scores to evaluate the soil erosion risk category (CORINE, 1992; Figure 2.13.4). A relative ranking of soil erosion risk was obtained through the summation of individual erosion risk scores for each of the parameters soil susceptibility, rainfall, topography and land cover. Erodibility is estimated from soil texture, depth and stoniness, extracted from the soil map of the European Communities (CEC, 1985). Erosivity is estimated from the Fournier and Bagnouls–Gaussen climatic indices. Slope gradient is included, but without a slope length correction, and vegetation and crop management are collapsed into two categories, protected and not fully protected, using data from the associated CORINE land cover database. The vegetation and crop management factor is the most poorly parameterised factor. All the factors are combined to estimate three categories of potential and actual soil erosion risk. Potential risk excludes vegetation factors, and so identifies land at risk, whereas actual risk includes the vegetation factor to indicate the protective influence provided by present land cover and the dangers inherent in land use changes. The resulting maps were produced at a 1-km resolution for southern Europe, excluding northern Europe. The area of land in this region with a high erosion risk totals 229 000 km^2 (about 10 % of the rural land surface).

The static CORINE approach relies heavily on risk assessment by experts, and it remains difficult to evaluate the effect of changes in land use and/or climate on the erosion risk as no quantitative estimate of soil erosion is made. For the same reasons, it is not feasible to incorporate more detailed data, nor is it possible to evaluate the accuracy of the final result.

2.13.3.5 The USLE/ESB Approach

The European Soil Bureau (ESB) initiated a project that aimed to assess erosion risk at a continental level on the basis of European datasets available. The end result is a set of maps that can help identify regions that are susceptible to rill and inter-rill erosion (van der Knijff *et al.*, 2000).

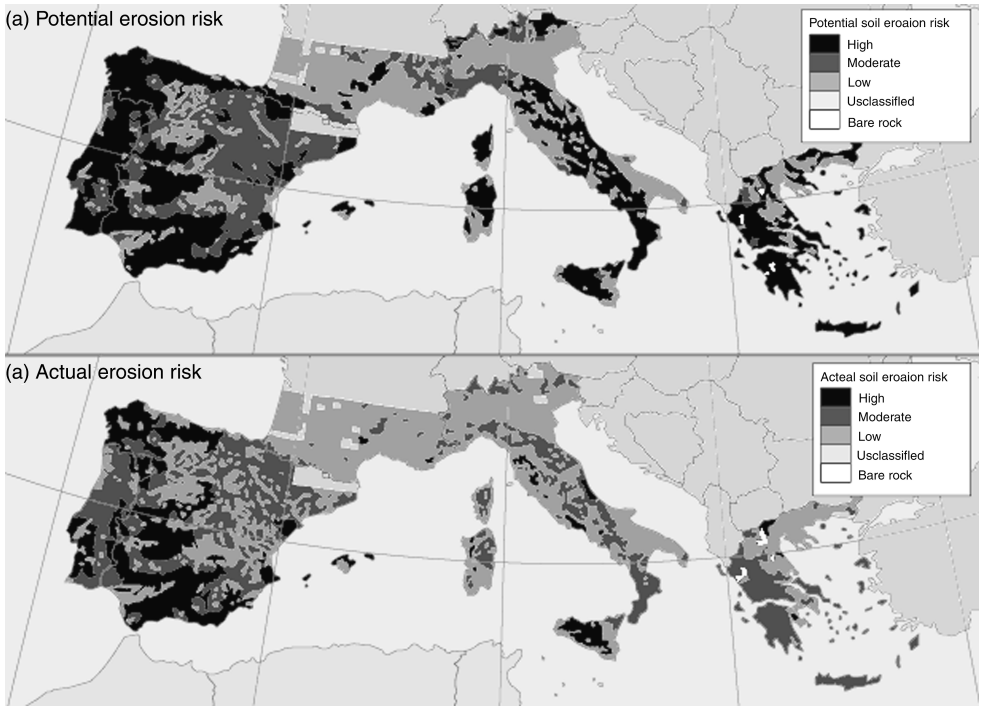

Figure 2.13.4 (a) Potential and (b) actual erosion risk as estimated by the CORINE methodology (Figure adapted from the original maps in Soil Erosion Risk and Important Land Resources in the Southern Regions of the European Community EUR 13233EN. © European Communities, 1992, reproduced with permission)

The method uses the USLE in an attempt to produce a comprehensive pan-European soil erosion risk assessment map at 1-km resolution (Figure 2.13.5). The rainfall erosivity factor was estimated using linear regression equations for northern and southern Europe, allowing for a smooth transition between the two. The soil erodibility factor was calculated using an exponential equation based on the geometric mean weight diameter of the primary soil particles, derived from surface texture composition as presented in the European Geographical Soil Database for Europe (Heineke *et al.*, 1998). The slope length factor was assumed to be constant, whereas the slope factor was derived from the slope gradient of a 1-km resolution European-wide Digital Elevation Model. The land cover factor was estimated using an exponential scaling function linked to NDVI from NOAA AVHRR. The visual appearance of the map was improved using a filter that replaces the actual pixel values by the median of all pixel values within a 5-km search radius.

The approach represents an application of the USLE methodology, with the benefit of incorporating improved and updated data layers once they become available at a pan-European scale. Unlike for the USA, where there exists a large database, lack of systematic data across the different agro-environments of Europe will make it very difficult to validate the methodology and resulting map. Moreover, the application of the method to mountainous regions is Europe is questionable.

2.13.3.6 The INRA Approach

The approach elaborated by INRA (Institut National de la Recherche Agronomique, France) presents a factorial approach.

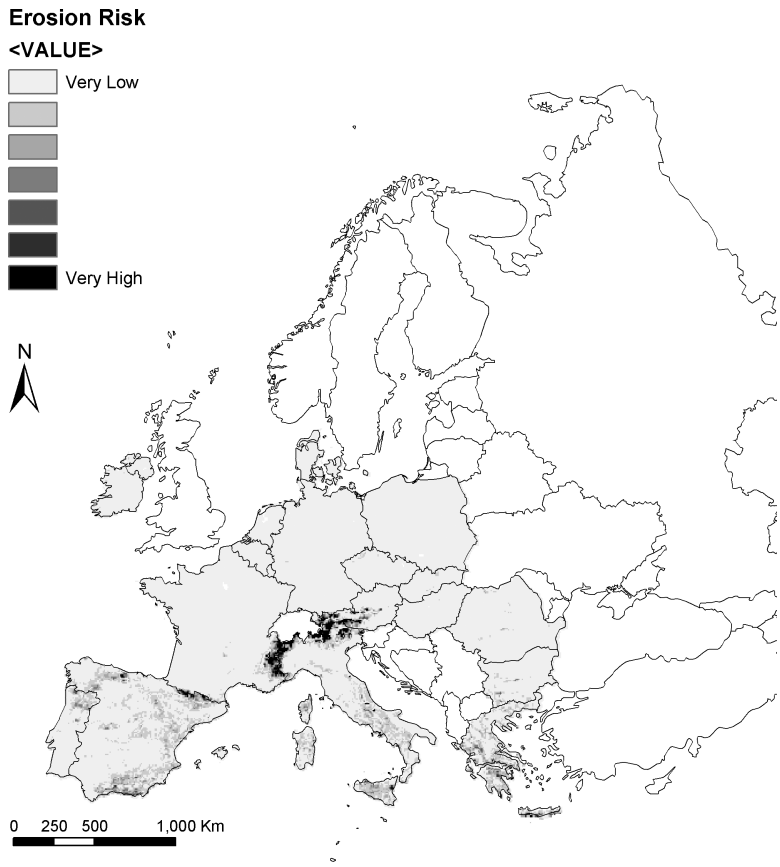

Figure 2.13.5 Actual erosion risk as estimated by the ESB/USLE approach (modified from Soil Erosion Risk Assessment in Europe by Van Der Knijff *et al*., 2000, EUR 19044 EN, reproduced with permission)

The INRA approach uses a hierarchical multi-factorial classification designed to assess average seasonal erosion risk at a regional scale (Le Bissonnais *et al.*, 2001). The annual soil erosion risk for Europe is based on empirical rules that combine data on land use from the CORINE Land Cover database, soil crusting susceptibility, soil erodibility (determined by pedotransfer rules from the European Soil Database at a scale of 1 : 1 000 000; Le Bissonnais and Daroussin, 2001), relief (1 km × 1 km resolution) and meteorological data (Figure 2.13.6).

The INRA approach is simple and versatile, not requiring parameters that are not available at the national scale. The INRA approach is qualitative with the final information provided on a five-class scale of risk. Since the classes are not linked to quantitative values of erosion, it is impossible to assess the errors associated with the results.

2.13.3.7 The PESERA Approach

The Pan-European Soil Erosion Risk Assessment (PESERA) Project, an EU-funded 5th framework project, was initiated to develop and evaluate a physically based and spatially distributed model to quantify soil erosion in a nested strategy of focusing on environmentally sensitive areas relevant to a European scale.

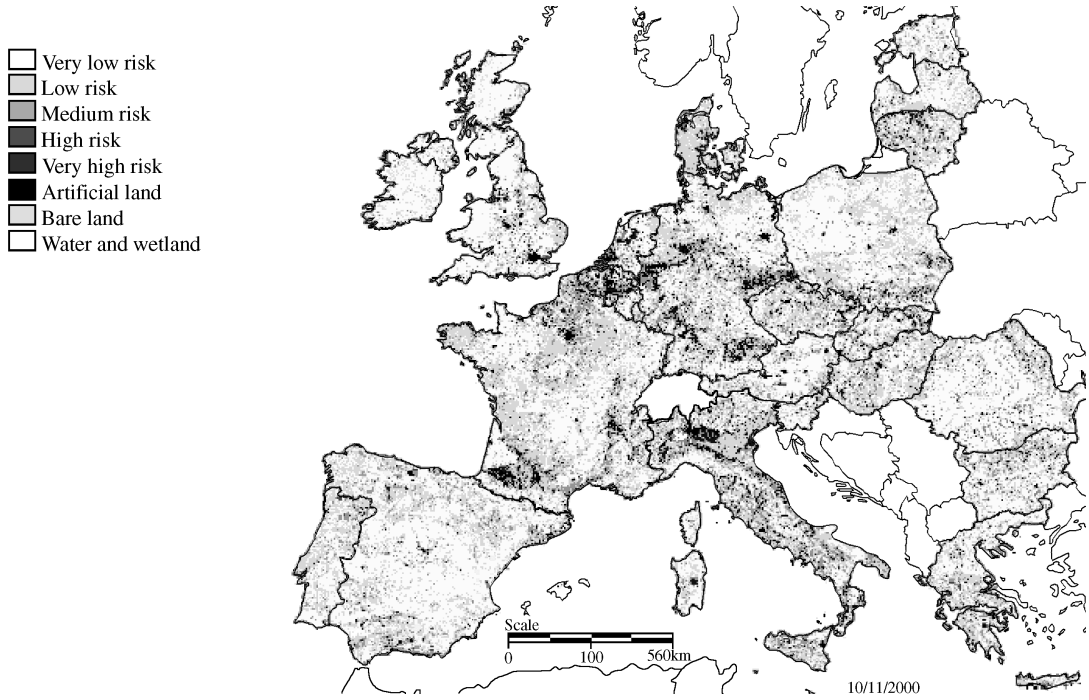

Figure 2.13.6 Annual soil erosion risk as inferred from soil sensitivity to erosion and rainfall erosivity (Soil Erosion Risk Assessment in Europe data are owned by the European Commission and were calculated by INRA, according to a methodology which was developed and is owned by INRA and to which reference can be made by citing the following publication: Le Bissonnais Y., C. Montier, M. Jamagne, J. Daroussin, D. King (2002). Mapping erosion risk for cultivated soil in France. Catena, 46, 207–220)

The PESERA model calculates expected mean erosion rates at 1-km resolution for the full range of environments in Europe (Figure 2.13.7). The model makes use of topography, soil, climate, land use and land management data to estimate ground cover, surface crusting, runoff and sediment transport and to provide an estimate of water and sediment delivered to stream channels. The estimates are consistent with finer scale erosion models for flow strips, evaluated at the slope base, and are integrated across the frequency distribution of storm magnitudes. The model is based on a partition of daily precipitation into Hortonian and saturation overland flow, subsurface flow and evapotranspiration. Hortonian overland flow, which is mainly responsible for soil erosion, is generated with respect to local soil and subsurface moisture characteristics. Some allowance is also made for snow accumulation and melting. Model forecasts are calibrated against runoff plots and small catchment data. The model output is also compared with other assessment methods at different resolutions and across different agro-ecological zones (Gobin and Govers, 2003). Detailed reports on these efforts are available from the Project's website.

The major advantage of the PESERA approach towards Europe-wide soil erosion assessment is the application of a process model that can be used for validation at high resolutions and for Europe-wide forecasting at a coarse resolution, so that cross-scale reconciliation is as explicit as possible. The PESERA model produces a quantitative forecast of soil erosion and soil cover, offering great promise for scenario analysis and impact assessment. It is now being demonstrated that it has the potential to respond explicitly and rationally to changes in climate or land use.

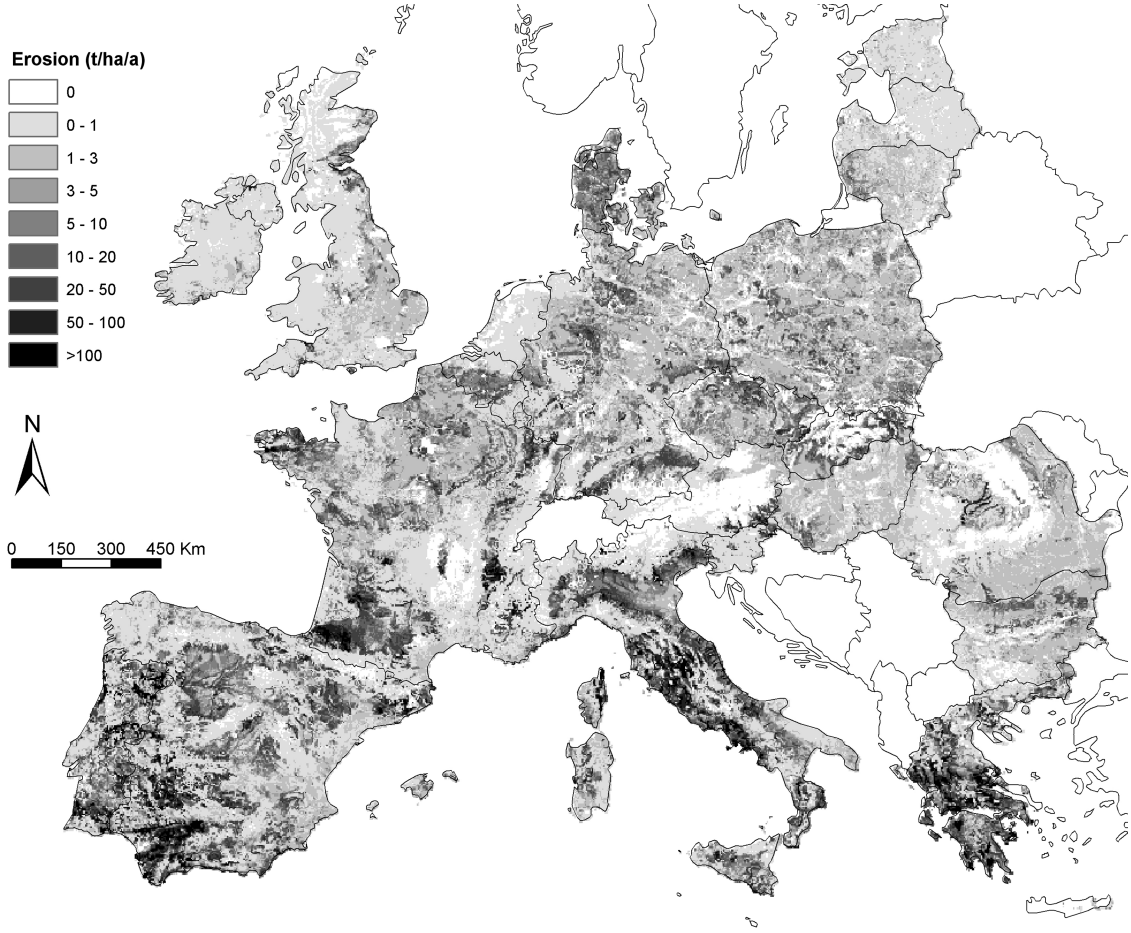

Figure 2.13.7 Preliminary results of the PESERA model across Europe (modified from Gobin and Govers, 2003, reproduced with permission)

2.13.4 DISCUSSION

Methods based on questionnaire surveys (GLASOD map; Oldeman *et al.*, 1991) or erosion measurement sites (Hot Spots map; Turner *et al*., 2001) are inadequate on their own. Estimates of the area affected by actual soil erosion at regional and national levels are not readily available, because measurements are difficult and usually expensive to make. Soil erosion often takes place over long periods before the true extent is appreciated and long-term accurate data are scarce. The temporal and spatial patchiness of soil erosion makes interpolations between limited available data scientifically not justified. Differences between expert assessments and measurement methods reduce the comparability between the limited data available even further.

Factorial scoring methods such as the static CORINE approach rely heavily on risk assessment by experts, and it remains difficult to evaluate the effect of changes in land use and/or climate on the erosion risk as no quantitative estimate of soil erosion is made. For the same reasons, it is not feasible to incorporate more detailed data, nor is it possible to evaluate the accuracy of the final result. In addition, the sharp delineation of

source data in both the geographic and attribute space inevitably results in information loss, and the results depend strongly on the class limits and the number of classes used.

There exists a continuous spectrum between mapping based on ranked indicators and empirical models such as the (R)USLE. The potential risk calculations are based on climatic, topographic and edaphic conditions, whereas the actual risk takes into account present land cover and land use. In many ways, these empirical models are comparable to the factorial scoring methods used for producing the CORINE maps. Adhering closer to the USLE are the RIVM maps at 50 km × 50 km resolution for Europe (RIVM, 1992) and the USLE/ESB map at 1 km × 1 km resolution (van der Knijff *et al.*, 2000). Although the USLE has been the most widely applied model, it is now considered as conceptually flawed. Despite this, the USLE-based methods have the immediate benefit of transferring distributed data into simple factors. However, any model based on the USLE will not include gully erosion, which, as Poesen *et al.* (2003) have shown, can be a major contributor to total erosion in (at least some parts of) Europe. As part of the ranked indicator methods, the INRA approach presents a more elaborated method, which was tested at high resolution (50 m) for significant areas and compared favourably with the PESERA modelling method (Gobin and Govers, 2003). The INRA method explicitly takes into account land use and the interactions particularly with soil sensitivity to erosion. However, limitations relate to the qualitative and non-physical nature of the method.

Actual levels of erosion are difficult to measure, so estimates, based on the available physical evidence provide necessary information on risk, rather than an actual occurrence, of erosion for policy and management purposes. Process modelling methods such as the PESERA approach offer great scope but may simplify the set of processes operating and may therefore not be appropriate under particular local circumstances. The explicit cross-scale reconciliation of the PESERA approach provides a method to test the coarse resolution model forecasts against finer resolution measurements. The PESERA method is the only approach that explicitly incorporates the effect of vegetation and ground cover at different steps in the model.

All of the methods presented require calibration and validation, although the type needed is different for each category. The advantage of quantitative and process modelling methods, such as PESERA, is that coarse-scale forecasts can be related to measured erosion rates so that explicit calibration and validation can be made with field monitoring data. Examples of validation and calibration using plot data and of validation using aggregated measurements (e.g. suspension load, pond sedimentation) or high-resolution maps have been produced within the framework of the PESERA Project (Gobin and Govers, 2003). However, aggregated measurements necessitate the use of concatenating different models to arrive at forecasts to be confronted with measurements (e.g. van Rompaey *et al.*, 2003); this in turn increases error propagation and introduces further uncertainties in the prediction. Despite the uncertainties involved in European-scale assessments of soil erosion, policy-makers need to know the area affected by soil erosion and an estimate of the magnitude at a regional and continental scale in order to formulate suitable remediation measures and mitigation strategies focussing on environmentally sensitive areas (Gobin *et al.*, 2004).

2.13.5 CONCLUSIONS

In order to formulate a European soil protection policy, pan-European soil erosion assessment methods should provide information of both the extent and the severity of the problem at the European scale as a first crucial step in a nested strategy of focusing on environmentally sensitive areas, which may require remedial measures to be taken (Gobin *et al.*, 2003, 2004). There is a huge difference between measured erosion, actual erosion risk and potential erosion risk. Certain factors may affect the risk of soil erosion, but they may not affect soil erosion in itself at present.

Runoff is the most important direct driver of severe soil erosion. Processes that influence runoff must therefore play an important role in any analysis of soil erosion intensity, and measures that reduce runoff are

critical to effective soil conservation. Any approach that explicitly takes into account the land use and management factors should be of potential interest to the policy-maker since these are the only factors at hand to control the erosion problem. It is apparent that the majority of approaches discussed in this chapter do not pay enough attention to land use and land management factors.

The temporal and spatial patchiness of soil erosion favours a risk analysis approach in order to make comparisons between regions and to complement field measurements and observations. However, modelling efforts should be thoroughly validated against erosion measurements, and a clear distinction should be made between modelled erosion risk and present-day erosion rates. A programme to monitor soil erosion across different agro-ecological regions and under different land uses should underpin both field observation/mapping exercises and regional soil erosion risk assessment methods. Only then is a sound approach ensured of estimations and mapping features that are directly validated and compared with measurements. Moreover, measuring campaigns may lead to new insights and therefore to better mapping and risk assessments.

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2.14

Assessing the Modified Fournier Index and the Precipitation Concentration Index for Some European Countries

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2.14.1 INTRODUCTION

In spite of shortcomings, criticisms and misuses, attempts are still made in many countries, including Europe, to assess the soil erosion risk based on the principles defined in the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978). The USLE was developed as a means of computing field-scale assessments of soil losses by rainfall from agricultural land and a rainfall/runoff erosivity index, R, was introduced to relate those field soil losses with rainfall, making the erodibility index, K, dependent on R. Also the RUSLE (Revised Universal Soil Loss Equation) (Renard *et al*., 1997) uses rain kinetic energy (KE) as (part of) the rain erosivity index.

Although progress has been made in obtaining more accurate values of rain intensities for short periods of time, the mean annual erosivity factor, $R = EI_{30}$, is still difficult to obtain. The reason is that the kinetic energy E and the maximum intensity I_{30} of the individual rainstorms not only need to be analyzed for a sufficient number of years, but also in many cases still insufficient rainfall records, such as rain intensity, but also and especially the correct, direct and continuously measured rain energy and raindrop size distribution during a rainstorm, are not available to calculate a rain erosivity index countrywide (Salles *et al*., 2000, 2002).

The rain kinetic energy is calculated from raindrop size distributions and related raindrop terminal fall velocity. With the exception of the study of Doelling *et al*. (1998), reporting seven years of drop size distribution measurements in northern Germany, we are not aware that more continuous data in space and time

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of drop size distributions are available. Also, devices that allow direct measurement of rain kinetic energy (Madden *et al*., 1998; Jayawardena and Rezaur, 2000) are still in the stage of development. Moreover, in most of the studies the energy is calculated for rains with a vertical downward trajectory; the important effect of wind-driven rains on the energy produced is in most (if not all) cases neglected.

With those sporadic available drop size measurements, empirical relationships were established between rain kinetic energy (*KE*) and rain intensity (*I*). Salles *et al*. (2002) discussed two expressions for *KE*: (1) the rain kinetic energy expressed per volume of rain or volume-specific kinetic energy, and (2) the rain kinetic energy rate or time-specific kinetic energy. One is the empirical log-linear equation established by Wischmeier and Smith (1958) based on data of Laws (1941), Gunn and Kinzer (1949) and Laws and Parsons (1943):

$$KE = a + b\log I \tag{2.14.1}$$

where a and b are constants derived through the regression.

Although this relationship is still widely used, it should be mentioned that Wischmeier and Smith (1958) did not provide any indication on the physical basis of this log-linear equation (Salles *et al*., 2002). The kinetic energy formulation of Wischmeier and Smith (1958) was revised by Renard *et al*. (1997) for establishing the RUSLE. In most countries outside the USA, the rainfall energy has been and still is calculated using a regression between energy and rainfall intensity established for US conditions (Renard *et al*., 1997). Not validating this regression for climatic conditions outside the USA certainly leads to doubtful energy and erosivity values.

Polynomial equations were also tested by Renard (1983) on kinetic energy derived from drop size distributions measured in Belgium. A linear relation gave the best regression, although Govers (1991) reported unrealistically high kinetic energy values when applying the equation proposed by Bollinne *et al*. (1984).

Homerin and Renard (1983) found a relationship between EI_{30} and rain intensity for European conditions. This relationship is different from the USLE erosivity index (Wischmeier and Smith, 1978) and from the *R* index used in the RUSLE (Renard *et al*., 1997). Bollinne *et al*. (1979) calculated the USLE *R* index for four meteorological stations in Belgium. However, they found low correlations with field soil losses. A better correlation was found when only rainfalls higher than the threshold value of 1.27 mm were considered (Bollinne, 1982).

These constraints led to a search for alternative procedures to assess the rain aggressivity from more easily determinable rainfall parameters. Arnoldus (1980) proposed the Modified Fournier Index (MFI) considering monthly and yearly total rainfall amounts over a number of successive years. Bollinne *et al*. (1980) tried to correlate the MFI with calculated EI_{30} values. Bergsma (1980) calculated EI_{30} values for the Netherlands according to the Wischmeier (1962) method based on annual rainfall, maximum 1 h rainfall with a 2-year return period and maximum 24 h rainfall with a 2-year return period. From 4 years of field experiments in Germany, Richter (1979, 1984) found differences in erosivity between the strong convective summer rains and the frequent advective winter rains. Bader and Schwertmann (1980) and Rogler and Schwertmann (1981) calculated erosivity values of individual rainstorms in Bavaria, Germany, as a function of the total rainfall amount of the rainstorm and I_{30} being the maximum 30-min intensity. Other relationships between the *R* index and the mean annual rainfall or the mean summer rainfall in Germany were established by Saupe (1985), Hartmann (1986), Hirche (1990), Mollenhauer *et al*. (1990) and Botschek (1991).

Pihan (1986) made an erosivity map of France also using calculated EI_{30} values. For a small region such as the Alsace in France, with three meteorological stations, Strauss *et al*. (1997) found a correlation between the total annual or summer rainfall and the calculated EI_{30}.

Once again it should be stressed that only in a few cases were the so-called (calculated) erosivity values validated against measured soil losses from field experiments.

The aim of this chapter is to attempt to propose a methodology for calculating rainfall indices based on monthly rainfall amounts for their possible use in relation to describing and assessing the erosive potential of rainfall in view of their validation with field erosion losses in different countries and regions in Europe.

2.14.2 THE MODIFIED FOURNIER INDEX (*MFI*)

Fournier (1960) developed an index, the *Fournier Index* (*FI*), for correlation with sediment loads in rivers of large catchments:

$$FI = {p_{\max}}^2/P(\text{mm}) \tag{2.14.2}$$

where $p_{\max}$ is the mean monthly rainfall amount (mm) of the wettest month of the year and P is the mean annual rainfall amount (mm).

However this *FI* has shortcomings as an estimator of the rain erosivity. As low amounts of rainfall also have erosive power, an increase in total rainfall amount should result in an increase of erosivity. It is not logical that if the maximum monthly rainfall $p_{\max}$ remains the same with the mean annual rainfall P increasing, *FI* decreases.

Therefore, Arnoldus (1980) modified *FI* to give a *Modified Fournier Index* (*MFI*) considering the rainfall amounts of all months in the year:

$$MFI = \Sigma p^2/P(\text{mm}) \tag{2.14.3}$$

where p is the monthly rainfall amount (mm) and P is the annual rainfall amount (mm).

Constituted by the European Council Decision of June 1985, the CORINE Soil Erosion Risk and Important Land Resources Project was carried out by a group of European experts. A specific topic of prime importance for the European Community policy was the determination of the erosion risk in the Mediterranean region. The assessment of the erosion risk (potential and actual) was also based on the principles and the parameters defined in the USLE (CORINE, 1992).

In the CORINE project, the decision was taken to use the product of two indices as a basis for calculating a so-called 'erosivity index': a combination of the *MFI* and the Gaussen – Bagnouls aridity index (*BGI*) (Gaussen, 1963). The *MFI* was calculated from the mean monthly and mean annual rainfall amounts and classified as shown in Table 2.14.1.

Although the *MFI* gives an adequate measure of rainfall variability, it does not take into account the general aridity of the climate, and hence the likelihood of short-period intense storms during otherwise dry seasons. In order to strengthen the assessment of erosivity, CORINE included a second climatic index: the

TABLE 2.14.1 Classification of *MFI*

MFI range	Description	Class
<60	Very low	1
60–90	Low	2
90–120	Moderate	3
120–160	High	4
>160	Very high	5

TABLE 2.14.2 Classification of *GBI*

GBI range	Description	Class
0	Humid	1
0–50	Moist	2
50–130	Moderate	3
130–160	Dry	4
>160	Very dry	5

Gaussen–Bagnouls aridity index (*GBI*) (Gaussen, 1963), based on the mean monthly temperature and the mean monthly rainfall amounts:

$$GBI = \sum_{i=1}^{12} (2t_i - p_i)k_i \tag{2.14.4}$$

where t_i is the mean temperature for the month i (°C), p_i is the total rainfall amount of the month i (mm) and k_i is the proportion of the month during which $(2t_i - p_i) > 0$. The *GBI* is classified as shown in Table 2.14.2.

The two classes of the climatic indices were combined to give a so-called 'erosivity index' (a relative number) as follows:

$$\text{erosivity index} = (MFI)_{\text{class}} \times (GBI)_{\text{class}}$$

It should be mentioned that the combination of two indices or two classes does not imply a scientific meaning or a validation or relation of the index with soil losses from field plots. However, it stresses the importance of taking a climatic (aridity) factor into account to delineate the zones of application. Use was made of this index to make the climatic erosivity map of southern Europe (CORINE, 1992) with regard to the potential and actual erosion risk assessment.

2.14.3 ASSESSMENT OF THE MODIFIED FOURNIER INDEX (*MFI*) FOR 16 EUROPEAN COUNTRIES

In the frame of a Task Force of the European Society for Soil Conservation (ESSC), an attempt was made by Vermeulen (1999) to establish a rainfall network of 16 countries in Europe: Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, the Netherlands, Norway, Portugal, Spain, the UK and Sweden. The persons who provided rainfall data are acknowledged at the end of this chapter. However, we are also aware that since that time other studies on assessment of rain erosivity or rainfall aggressivity have been carried out on a regional, national or continental scale, with or without validation against field soil losses from erosion.

Vermeulen (1999) collected monthly rainfall data for at least 10 successive years. This procedure was based on previous research on rain erosivity and precipitation concentration in Spain and Belgium (Michiels *et al.*, 1992). If monthly data for several successive years were not reported or not available, the average monthly rainfall data were taken into account for the calculation of the *MFI*.

According to the available data sets, two different procedures can be followed to calculate *MFI*. In the first procedure the monthly rainfall amounts are averaged over a number of years. The *MFI* is then calculated from this averaged rainfall data set and reported as $(MFI)_1$. In the second procedure, the *MFI* is calculated from the

TABLE 2.14.3 Number of stations providing rainfall data from 16 European countries

Country	Monthly data	Average monthly data	Stations	Surface (km^2)	Surface/station (km^2)
Denmark	5	68	73	43031	590
Belgium	34	3	37	30513	825
Spain	313	296	608	504879	830
Portugal	98	0	98	91641	935
Italy	285	35	320	301245	941
Greece	46	91	137	131944	963
The Netherlands	5	33	38	36624	964
Ireland	18	36	54	68893	1276
Luxembourg	2	0	2	2586	1293
Germany	6	233	239	356178	1490
UK	7	139	146	244022	1671
Austria	50	0	50	84000	1680
France	0	148	148	543998	3676
Finland	31	13	44	337000	7659
Sweden	43	10	53	449750	8486
Norway	16	20	36	324000	9000
EUROPE	**959**	**1125**	**2083**	**3550**	**1704**

monthly rainfall amounts of each individual year and the *MFI* averaged over a number of years. Those long-term average values are reported as $(MFI)_2$. Finally, the rainfall data were reported from 2083 measuring stations spread over an area in Europe of 3 550 304 km^2 (Table 2.14.3) between 10.75° and 30.35° latitude and between 35.00° and 69.65° longitude.

A total of 959 stations allowed the calculation of $(MFI)_2$ and also $(MFI)_1$, and from the 2083 stations only $(MFI)_1$ could be calculated (Vermeulen, 1999).

In general, the average monthly rainfall data resulted in lower $(MFI)_1$ values compared with the $(MFI)_2$ values calculated from monthly values for at least 10 successive years. As the highest *MFI* values should be used for establishing an erosivity index, attempts were made to use the $(MFI)_2$ values for this purpose.

Although rainfall and hence rain distribution, rain erosivity and rain aggressivity are not restricted within the individual country boundaries, an attempt was made to find relationships between $(MFI)_1$ and $(MFI)_2$ for each country.

Table 2.14.4 gives the coefficients (a, b, r^2) of the relationship between $(MFI)_1$ and $(MFI)_2$:

$$(MFI)_2 = a(MFI)_1{}^b \tag{2.14.5}$$

Although *MFI* is not limited to country boundaries, the European countries were put into three zones
Northern and mid-Europe:

80 % of the stations have $(MFI)_2$ values between 60 and 160 mm:

$$(MFI)_2 = 1.463(MFI)_1{}^{0.9739} \text{ with } r^2 = 0.955 \tag{2.14.6}$$

Scandinavia:

80 % of the stations have $(MFI)_2$ values between 0 and 90 mm:

$$(MFI)_2 = 2.6001(MFI)_1{}^{0.8696} \text{ with } r^2 = 0.948 \tag{2.14.7}$$

TABLE 2.14.4 Relationship between $(MFI)_1$ and $(MFI)_2$

Country	No. of stations	a	b	r^2
Austria	49	1.46	0.95	0.99
Belgium	33	1.49	0.95	0.99
Denmark	4	1.50	0.95	0.99
Finland	30	1.29	0.98	0.97
Germany	5	1.18	1.00	0.99
Greece	45	2.25	0.89	0.96
Ireland	17	1.67	0.92	0.99
Italy	283	1.72	0.95	0.96
Norway	12	1.28	1.00	0.99
Portugal	97	2.02	0.93	0.99
Spain	244	2.56	0.87	0.96
Sweden	42	1.45	0.97	0.99
Europe	**885**	**1.80**	**0.94**	**0.94**

Southern Europe:

80 % of the stations have $(MFI)_2$ values between 90 and >160 mm (with values up to 400 mm):

$$(MFI)_2 = 1.924(MFI)_1^{0.9197} \text{ with } r^2 = 0.995 \tag{2.14.8}$$

2.14.4 ASSESSMENT OF THE PRECIPITATION CONCENTRATION INDEX *PCI*

In an attempt to define temporal aspects of the rainfall distribution within a year, Oliver (1980) proposed the Precipitation Concentration Index *PCI*, derived from the Index of Employment Diversification (Gibbs and Martin, 1962):

$$PCI = 100\Sigma p^2/P^2(\text{mm}) \tag{2.14.9}$$

where p is the rainfall amount of each month of the year (mm) and P is the yearly rainfall amount (mm).

The theoretical limits of the *PCI* are obtained as follows:

- when the rainfall in each month of the year is the same, the *PCI* equals 8.3;
- when all the rainfall of the year occurs in a single month, the *PCI* equals 100.

The *PCI* permits grouping of data sets according to the derived value. Oliver (1980) came to the conclusion that a *PCI* of <10 suggests a uniform distribution, a value from 11 to 15 denotes a moderate seasonal distribution and a value from 16 to 20 denotes a seasonal distribution. An index above 20 represents strong seasonal effects, with increasing values indicating increasing monthly rainfall concentration (Table 2.14.5).

According to the available data sets, two different procedures were followed for calculating the *PCI*. In the first procedure, the mean monthly rainfall amount is estimated by averaging the monthly rainfall data over a

TABLE 2.14.5 Classification of *PCI*

PCI range	Description
8.3–10	Uniform
10–15	Moderate seasonal
15–20	Seasonal
20–50	Strong seasonal
50–100	Isolated

number of years. Subsequently, the *PCI* is calculated from this estimated mean rainfall data set. These values will be reported as PCI_1. In the second procedure, the *PCI* is calculated from the monthly rainfall amounts of individual years and these annual *PCI* values are averaged over a number of years. These values are reported as PCI_2.

The first procedure results in *PCI* of an average year if the series of years is long enough. The second procedure will yield long-term average *PCI* values of individual years.

The following relation exists between PCI_2 and PCI_1 for 885 stations in Europe:

$$PCI_2 = 0.7129 {PCI_1}^{1.2869} \text{ with } r^2 = 0.638 \tag{2.14.10}$$

The rainfall in the southern countries of Europe is concentrated with strong seasonal zones in southern Spain, Portugal and Crete. In central Europe, the rainfall is more uniform.

Although *MFI* and *PCI* are related by a factor $1/P$ (where P is the annual rainfall amount), no significant correlation could be found between $(MFI)_1$ and PCI_1:

For 2005 stations in Europe:

$$(MFI)_1 = 24.08 {PCI_1}^{0.52} \text{ with } r^2 = 0.028 \tag{2.14.11}$$

For 885 European values:

$$(MFI)_2 = 87.95 {PCI_2}^{0.09} \text{ with } r^2 = 0.002 \tag{2.14.12}$$

This means that the precipitation concentration is not in a direct relation with rainfall aggressivity. High monthly rainfall amounts (assumed to correspond to high erosivity values) and low monthly rainfall amounts (assumed to correspond to higher aridity) can be concentrated or uniformly distributed. This can explain the use of an aridity index (Bagnouls–Gaussen Index) (Gaussen, 1963) in the CORINE method (CORINE, 1992) to determine rainfall aggressivity classes.

2.14.5 RECOMMENDATION

It should be noted but also recommended that when dealing with the calculation of climatic indices, be it for erosion or assessment of aridity, care should be taken that regressions or relationships developed between climatic factors, measured in a particular climatological region, with a defined rainfall distribution and concentration, are in principle only to be used and applied within those zones with similar climatic conditions.

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2.15

Pan-European Soil Erodibility Assessment

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2.15.1 INTRODUCTION

Knowledge of soil erodibility is crucial for assessing the susceptibility of soil to erosion and, as such, is one of the key parameters in soil erosion modelling. However, it is difficult to define it precisely. It has to do with overland flow production in addition to detachment of soil material. The concept is described more or less restrictively by various authors (Bergsma, 1996), depending on the incorporation of other erosion affecting parameters such as rain, runoff, relief or plant cover. The reason for this confusion is clearly due to the strong interactions between these parameters. In this chapter, we consider soil erodibility as an intrinsic property of soil and we define it as the soil's susceptibility to be detached and transported by the action of raindrops and/or runoff (Bryan, 1968). Soil properties affecting erodibility are similar to those for crusting (texture, stoniness, aggregate stability, structure and shear strength) (Chapter 2.3), although the individual relationships between these properties and crusting and erodibility are different.

In assessing soil erodibility at a regional scale, we are confronted by two main difficulties: (i) the data availability and (ii) the spatial and temporal variability of soil surface characteristics (this will be discussed in the second part of the chapter). The most widely used method for the estimation of erodibility is the soil erodibility nomograph based on the study of the relationship between the measured K index of the Universal Soil Loss Equation (USLE) and soil properties (Wischmeier and Smith, 1978). It uses soil texture (silt plus fine sand and coarse sand content), organic matter content, plus soil structure of the topsoil and drainage class of the profile. However, it is widely recognised that very sandy and very clayey soils, and also crusting soils or soils with sesquioxides, can present problems when using the nomograph.

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Many more difficulties are encountered when trying to assess erodibility from soil maps at a broader scale. The European Geographic Soils Database (Le Bas *et al*., 1998) is currently the best and main source of distributed information on soil to assess erodibility and soil erosion-related parameters at the European level. However, the original scale of information is 1:1 000 000 and the information quality is variable between countries. Previous work at the same scale includes the CORINE erosion project (CEC, 1992). In this project, the erodibility factor was defined on the basis of the combination of soil texture (three classes), soil depth (three classes), and stoniness (two classes) that were available within the European Geographic Soils Database at a scale of 1:1 000 000. For each soil mapping unit, erodibility was calculated as a weighted product of the three parameters class values for the various taxonomic units included in the mapping unit. However, it was recognised that the assessment of erodibility needed further improvement, particularly in order to include more explicit factors such as organic matter and carbonate content. The need to use a soil map at a more detailed scale, e.g. 1:250 000, was also stressed (CEC, 1992). More recently, Kirkby *et al*. (2000) developed a more physically based approach for soil erosion risk mapping at European scale, in which an erodibility parameter was obtained with the approach presented here (Le Bissonnais *et al*., 2005).

The objective of this study is to derive soil erodibility from soil parameters available in the European Soil Database and to give related information on the accuracy of this parameter. The limits to the approach are also discussed.

2.15.2 DEFINING PEDO-TRANSFER RULES FROM THE EUROPEAN SOIL GEOGRAPHIC DATABASE TO ASSESS SOIL ERODIBILITY FOR EUROPE

2.15.2.1 Methods and Available Data

Erodibility was extracted from the European Soil Database using chained pedo-transfer rules (Bouma and Van Lanen, 1986; Daroussin and King, 1996). However, basic information necessary for estimating this parameter are not present in the soil database (organic carbon and clay content, clay mineralogy, cation content, etc.). In addition, soil mapping units are very heterogeneous at this scale, and dominant soil types may represent less than 50 % of the unit area in some cases. Therefore, pedo-transfer rules are kept simple and the resulting parameters are still approximate for typological units. Averaged values are given for each soil mapping unit (Figure 2.15.1).

2.15.2.2 Soil Physical and Chemical Properties Influencing Erodibility

There is a large amount of literature dealing with the effect of soil physical and chemical properties on aggregate stability and erodibility (Le Bissonnais, 1996; Amézketa, 1999). However, at the present stage, there is very little information available within the European Soil Database concerning these properties, except for soil texture, which is only very roughly described in five classes. Still the soil taxonomy used for soil classification at order level is based on pedologically fundamental soil characteristics. Our working hypothesis is that such physico-chemical characteristics can be inferred from soil name and combined with textural information, in order to assess soil erodibility.

The erodibility class is therefore based on the combination of a textural/parent material factor and a physico-chemical factor. These two intermediate factors are themselves established from pedo-transfer rules which are described hereafter.

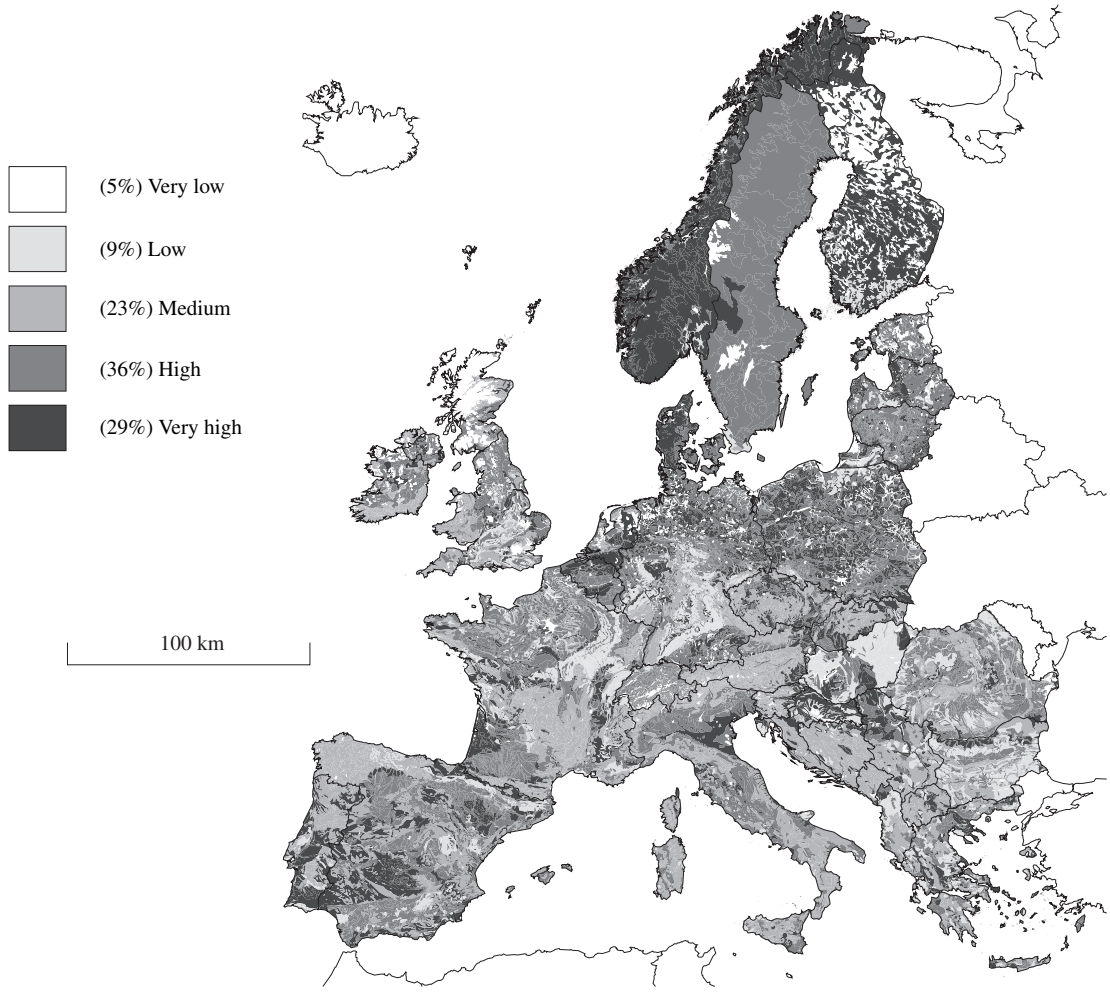

Figure 2.15.1 Textural factor of soil erodibility derived from the European Soil Database (values in parentheses indicate the proportion of Europe in each class)

2.15.2.3 Effect of Soil Texture on Erodibility

Because the dominant soil surface texture from the European Soil Database contains only very rough baseline information and as it does not allow discrimination between soils showing significant differences in texture with regard to effect on erodibility, we propose to refine this information by taking into account other available information. A specific parameter was therefore established and named 'textural factor of soil erodibility'.

The textural factor of soil erodibility is based on a combination of the dominant soil texture and the type of parent material. Although the influence of stoniness on erodibility has been shown (Poesen *et al*., 1994, 1999), the effect is highly variable, depending on several parameters data for which are not available at the European scale. Therefore, this parameter is not included in the pedo-transfer rule at the moment. Massive rocks such as

granite or limestone are classified as of low textural erodibility, whereas potentially loose rocks such as sand or molasses are classified as of high textural erodibility. Textural erodibility is also high for coarse and medium textures (Figure 2.15.1).

2.15.2.4 Effect of Soil Physico-chemical Properties on Erodibility

Based on the knowledge that in addition to the effect of texture, organic matter, Na cations, carbonates, Fe and Al oxides may increase or decrease erodibility, a physico-chemical factor of soil erodibility (five classes) is defined. It is derived from the soil name information (third level) by taking into account the positive or negative effect of organic matter content, carbonates, cations and other pedogenetic characteristics on structural properties.

Soil units are classified as follows:

- very low physico-chemical erodibility: Histosols;
- very high physico-chemical erodibility: Solonchak, Solonetz;
- low physico-chemical erodibility: Rendzina, Chernozem, Kastanozem, Greyzem, Phaeozem, Ferralsol;
- high physico-chemical erodibility: Podzoluvisols, Podzol, Arenosol, Andosol, Planosol, Xerosols;
- medium physico-chemical erodibility: Acrisols, Lithosols, Fluvisols, Regosols, Ranker, Vertisols and all other soils with the exceptions of (i) those which are dystric, gleyic, albic, planic, spodic, that are classified as high physico-chemical erodibility, and (ii) those which are calcaric, chromic, calcic, humic, that are classified as low physico-chemical erodibility (Figure 2.15.2); any improvement in the accuracy and resolution in the soil database would reduce the uncertainty of these two factors and would also allow refining of these pedo-transfer rules.

2.15.2.5 Combination of Physico-chemical and Textural Factors of Soil Erodibility

The second step is to combine the two intermediate erodibility factors. The rules are shown in Table 2.15.1. The final erodibility class takes the value of the textural erodibility factor if the class of the physico-chemical erodibility factor is medium; otherwise, it is decreased or increased depending of the value of the physico-chemical erodibility factor (Figure 2.15.3).

2.15.3 LIMITS OF THE PEDO-TRANSFER RULE METHOD AND FURTHER IMPROVEMENTS OF SOIL ERODIBILITY ASSESSMENT

2.15.3.1 Process-based Analysis of Soil Erodibility

In addition to the low accuracy and resolution of the data used, the principal limitation of the pedo-transfer rule method presented above is that it is not physically based. In fact, current conceptual models for soil erosion distinguish between inter-rill and rill erosion areas and divide soil erosion into sub-processes such as detachment, transport and sedimentation.

In the process-based model developed by Hairsine and Rose (1991), it has been supposed that soil detachment in inter-rill areas is due to raindrop impact and that the only role of overland flow is to transport the detached aggregates downslope. This model calculates the fragment-size distribution of material leaving a uniform area through the concept of settling velocity classes. It describes the rate of rainfall detachment and uses an empirical detachability parameter. We hypothesise that aggregate breakdown measurement can be used

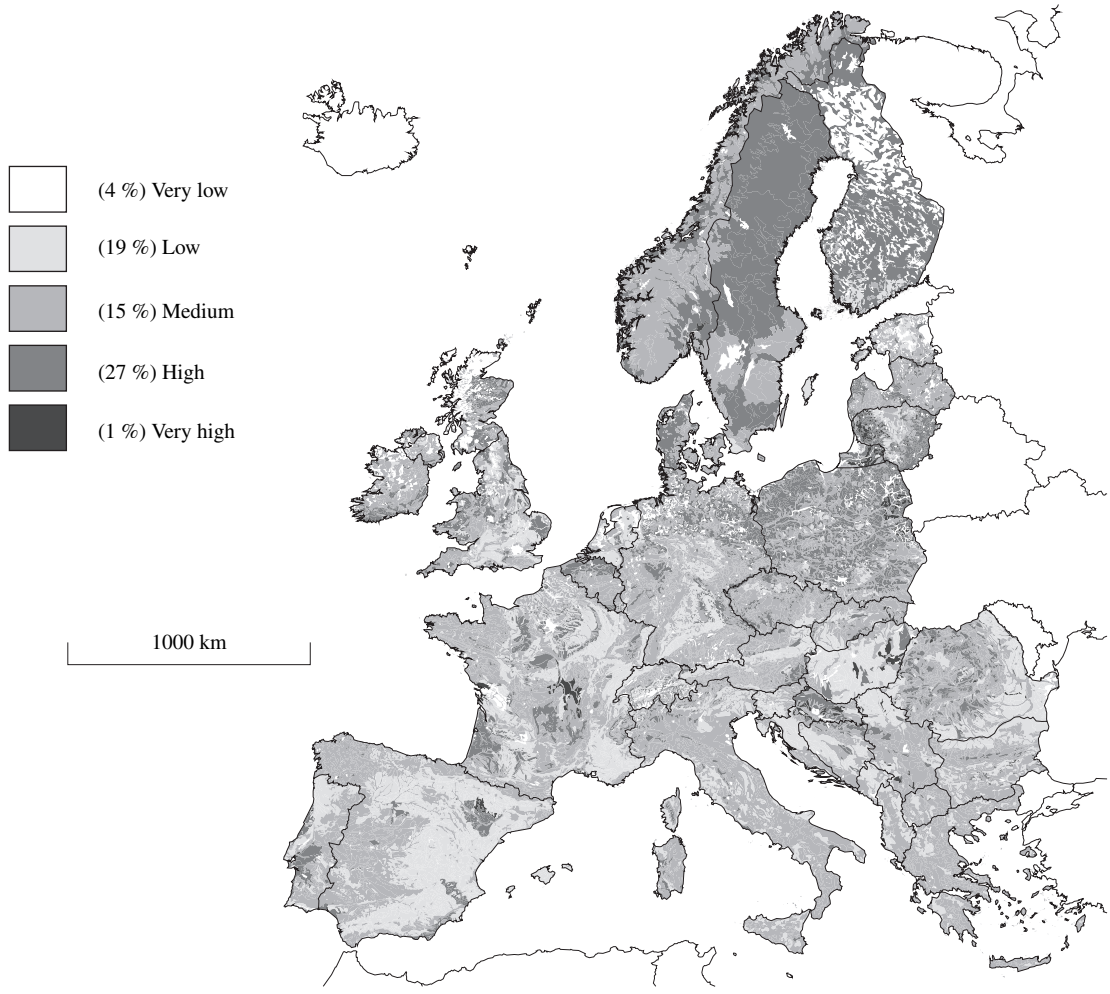

Figure 2.15.2 Physico-chemical factor of soil erodibility derived from the European Soil Database (values in parentheses indicate the proportion of Europe in each class)

TABLE 2.15.1 Assessment of soil erodibility by combination of physico-chemical and textural erodibility factors

	Physico-chemical erodibility				
	Textural erodibility				
	1 very low	2 low	3 medium	4 high	5 very high
1 very low	1	1	1	2	3
2 low	2	2	2	3	4
3 medium	3	3	3	4	5
4 high	3	4	4	4	5
5 very high	4	4	5	5	5

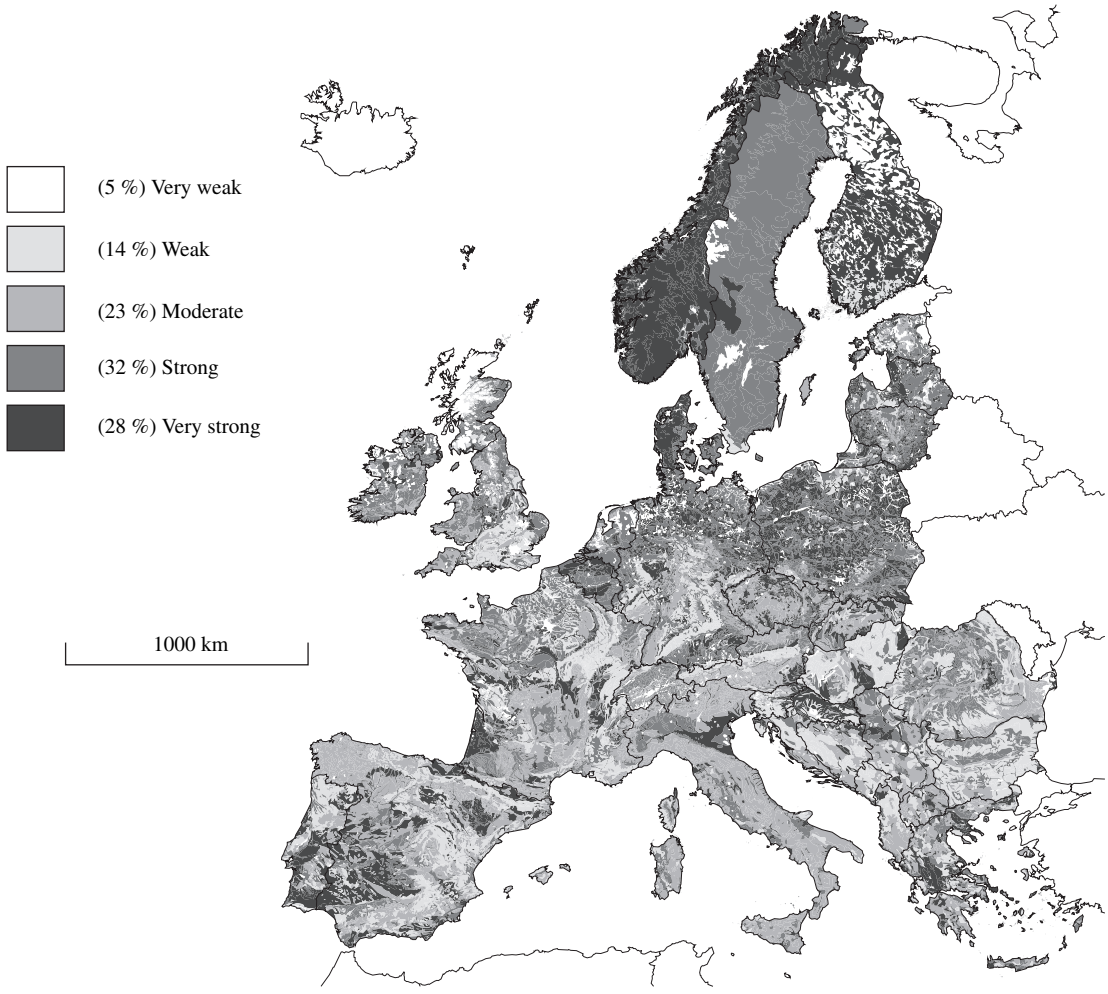

Figure 2.15.3 Soil erodibility class derived from the European Soil Database (values in parentheses indicate the proportion of Europe in each class)

to model the size distribution of fragments resulting from raindrop impact effects (Leguédois and Le Bissonnais, 2004). Aggregate breakdown is responsible for the production of micro-aggregates and particles, which are easily transported by splash and runoff. This effect of breakdown is particularly important in inter-rill erosion, where the runoff detachment capacity is limited.

On the other hand, when water concentrates, following pathways dictated by topography or by linear features associated with agricultural practices, discharge increases, which allows for detachment and transport by runoff, and eventually for rill and gully development. Soil erodibility for such concentrated flow conditions must be defined for each soil layer which may be affected by the incision process, and it can only be defined in relation to the erosivity of the flow, which is most often expressed using the bed shear stress. Soil erodibility is usually estimated from flow experiments in a flume, in which the bed shear stress is increased progressively by increasing discharge and/or slope, and where the sediment flux is measured at the outlet. The relationship

between bed shear stress and the sediment flux E is generally close to linear and expressed as

$$E = \begin{cases} 0 & \text{if } \tau \leq \tau_c \\ K_r(\tau - \tau_c) & \text{if } \tau > \tau_c \end{cases} \quad (2.15.1)$$

where τ is the bed shear stress, τ_c is the critical shear stress above which the sediment flux becomes significant and K_r is the increase in the sediment flux for a unit increase in shear stress. When the sediment flux is not measured and only the occurrence of rill incision is observed, only τ_c can be estimated. This relationship forms the basis of the rill and ephemeral gully erosion component of numerous soil erosion models, the parameters being considered as the erodibility characteristics of soil with regard to concentrated flow erosion.

It is clear from the complexity of the physical processes of inter-rill and rill erosion that the information available at the moment in the European Soil Database does not allow one to distinguish physically related inter-rill and rill erodibility parameters. However, the current research to determine these processes may allow the establishment, in the future, of separate empirical pedo-transfer functions between specific soil properties and inter-rill and rill erodibility parameters.

2.15.3.2 Temporal Variability of Soil Erodibility

The evolution of a soil surface during and between rainfall events, through rainfall, drying, freezing or biological effects, should be considered when studying erodibility. Its actual susceptibility to both inter-rill and rill erosion may change in time and become completely independent of aggregate stability measurement at a given moment. An important difficulty is that erodibility parameters are very sensitive to modifications of other factors such as soil structure, initial water content and vegetation development. The diversity of the determinants of erodibility makes predictions difficult: for example, we expect that no-tillage decreases erodibility owing to the combination of the effect of a higher initial crop cover, higher soil density (compaction is cumulative in such a cropping system) and maybe less opportunity for water table development near the surface, but we have no quantitative data on the evolution of erodibility.

In addition, under heavy rain, runoff may occur without the formation of a crust, and large aggregates and coarse fragments may be transported and eroded. Clearly, neither aggregate stability nor soil shear strength are good indicators of soil erodibility in this situation. Therefore, aggregate stability or mechanical resistance analyses are not sufficient to assess erosion risks in all circumstances.

2.15.3.3 Spatial Variability of Soil Erodibility

Erodibility may also vary in space at different scales within the field or landscape. Cultivation tends to homogenise soil surface characteristics, such as roughness or vegetation cover, at the field scale, but other factors, such as slope and soil texture, have their own spatial patterns at the scale of landscapes.

Spatial variability also exists at the local scale and may lead to high variability in erosion response, even for measurements on small plots for single rainfall events. Nearing *et al*. (1999) and Nearing (2000) showed, on the basis of numerous erosion plot experiments, that erosion measurements include large unexplained variability even between supposed replicates.

At the catchment scale, spatial heterogeneity will create in the landscape a mosaic of various hydrological and pedological units with different erosion responses. The connectivity between areas infiltrating or producing runoff plays an important role in flow concentration and erosion process when moving from plot to catchment scale. However, the current resolution of soil data at the European scale does not allow us to take into account this spatial variability. Therefore, averaged equivalent classes of erodibility are considered for larger soil units.

2.15.4 CONCLUSION

Because precise, accurate and extensive information about soil characteristics that would be required for soil erosion risk modelling and mapping are not yet available at a European level, pedo-transfer rules were developed based on available data. The European Soil Database provides a starting point for delineating crusting and erodibility at a European level. It is likely, during the next few years, that techniques to measure or derive data to quantify complex soil erodibility properties will improve. However, currently there is a serious lack of reliable soil runoff and erodibility measurements, essential for calibrating soil erosion risk models at a regional level in Europe. In the meantime, simple pedo-transfer rules and, in the immediate future, mathematically based pedo-transfer functions, offer the only alternative for estimating the soil component of erosion risk at the regional scale.

The approach described here could be useful as a 'first filter' in land evaluation studies to identify the limits for soil behaviour with respect to runoff and erosion. However, more precise assessment of erodibility, based on direct measurements, is needed to support policy making today.

Rill and inter-rill erosion on cultivated and natural soils results from aggregate breakdown and detachment of fragments by raindrops and runoff. The measurement of fragment-size distribution following breakdown and the analysis of aggregate breakdown mechanism allows for the assessment of the soil's susceptibility to erosion. It will help to replace the actual empirical erodibility parameters in erosion models by physically based soil-specific characteristics. The knowledge of the different aggregate breakdown mechanisms and also soil shear stress resistance will also allow the establishment of relationships between soil properties and erosion processes.

In addition, the relevance of erosion modelling, applied through a soil map at the 1:1 000 000 scale may be questionable. It is more appropriate at a scale of 1:250 000 or finer where real soil types are mapped, and soil/relief/climate/land-use interactions and physical processes are being evaluated. It is clear that the basic data to run such models at broad scales will be lacking for some parts of Europe for many years to come. In the absence of these data, the approach described in this chapter offers a chance of achieving results that can help in broad policy making in the near future.

ACKNOWLEDGEMENT

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2.16

Modelling Soil Erosion in Europe

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2.16.1 INTRODUCTION

The processes of soil erosion by water, wind and tillage result from the operation of the laws of physics and chemistry. Since these laws are presumed to be applicable throughout the universe (i.e. both within and outside Europe), and since all models of erosion represent an attempt to describe these erosional processes (albeit with emphases which vary from one model to another), one might expect there to be little that is unique about erosion modelling in Europe. However, this is not the case. Present-day erosion modelling in Europe differs notably from erosion modelling in the USA, for example in the much greater diversity of approaches. This European distinctiveness appears to be on the increase.

This chapter gives a brief survey of European erosion modelling and its scientific origins; its secondary aim is to describe, and to explore the reasons for, the above-mentioned distinctiveness. The focus here is on erosion by water. Wind erosion is not discussed, and tillage erosion barely mentioned. Also, the emphasis is on models which are useful for the practical business of soil conservation, although some models whose primary function is a greater understanding of the processes of erosion are also considered: this is mainly because such models may well have a role to play in the development of the next generation of models for soil conservation. Geomorphological models for long-term landscape evolution are, however, not discussed.

The first section of the chapter gives an overview of the application and development of erosion models in Europe. The second section lists erosion models which are currently used within Europe and provides a general overview of each model's aims and approach and the spatial and temporal scales to which it may be applied. The criteria for a model to be listed in this chapter are that it is actively maintained and used and basic information on its principles, structure and mode of operation is readily available. We recognise that this

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strategy may leave out some models (for example, those that are used by only a single person or group, and/or which are not widely advertised): we apologise here for any omissions. The final section focuses on some key issues regarding present-day model application and development (including data needs, model sensitivity, model validation and the need for calibration) and ends by speculating on the future of erosion modelling in Europe.

2.16.2 THE BACKGROUND TO SOIL EROSION MODELLING IN EUROPE

2.16.2.1 The Origins of European Erosion Modelling

Necessity is, to a very large extent, the mother of invention with regard to soil erosion models. The need for predictive models, an essential item in the soil conservationist's toolkit, was the driver for soil erosion modelling both in Europe and (at an earlier date) in the USA. This common motivation means that the story of European erosion modelling has strong synergies with the tale from across the Atlantic. However, there are also important differences.

2.16.2.1.1 The US Heritage

The origins of erosion modelling in Europe are strongly bound up with the beginning of erosion modelling in the USA. Since, however, some readers may be unfamiliar with the American story, we summarise it here.

Globally, the scientific subdiscipline of soil erosion modelling began in the USA in the middle decades of the twentieth century, perhaps 30–40 years before it began in Europe. The need for effective soil conservation in the USA first became apparent on a regional basis in the 1920s: it then rose to national prominence following the mid-western dust bowl erosion problems and as a result of the modernisation of agriculture throughout the USA with the consequent increase in the scale and intensity of agricultural activity.

Although individual instrumented erosion plots had been established as early as 1915, it was in 1928 that Hugh Hammond Bennett established a network of soil erosion experiment stations (Nicks, 1998). Data from plots at these stations were pooled, and from these data were developed a number of erosion models. At first these models were limited to consideration of a subset of the variables which influence erosion, or to a specific region (e.g. Cook, 1936; Ellison, 1945). Later, a model was developed with much wider potential application: the USLE (Wischmeier and Smith, 1978) (a list of acronyms is included at the end of this chapter). It is this notion of 'much wider' applicability, compared with earlier models, which led to the inclusion of the word 'universal' when naming the USLE (M Nearing, personal communication, 2001). Thus the name of the USLE is, for applications outside the USA, somewhat misleading and may have led to unrealistic expectations on the part of early users. This empirical model became the standard tool for soil conservationists in the USA, particularly in the prime agricultural areas of the mid-west, so much so, that its predictions eventually obtained legal force (e.g. Hauser, 1984). Later research focused on remedying some of the USLE's limitations, and this resulted in direct USLE derivatives such as MUSLE (Williams, 1975) and RUSLE (Renard *et al*., 1991; Yoder and Lown, 1995). Subsequent US erosion models, although more process-oriented, still incorporate elements of the USLE's approach to varying extents. Models such as ANSWERS (Beasley *et al*., 1980), EPIC (Williams *et al*., 1984); SWRRB (Williams *et al*., 1985), its descendant SWAT (Arnold *et al*., 1995) and AGNPS (Young *et al*., 1989) are strongly USLE based; CREAMS (Knisel, 1980), its descendant GLEAMS (Leonard *et al*., 1987) and KINEROS (Woolhiser *et al*., 1990), rather less so; while WEPP (Lane and Nearing, 1989; Flanagan and Nearing, 1995) is built upon a relatively minor USLE heritage.

Of course, the USA was not the only part of the world in which erosion models were developed. Locally specific empirical models, often conceptually similar to the USLE, were constructed in several places [e.g. the

African SLEMSA model by Stocking and Elwell (1973) and the Australian USLE-M by Kinnell (1997)]. A much more process-oriented approach was followed in one or two other models, such as the Australian GUEST model (Misra and Rose, 1996; Yu, 2003).

2.16.2.1.2 First Applications of Erosion Models in Europe

In Europe, by contrast, there was no single dramatic event, no single equivalent of the US dust bowl, to kick-start research into soil erosion. Instead – as is described in other chapters of this book – erosion was more gradually perceived to be a problem from the late 1970s onwards.

Under traditional low-intensity agriculture, European soil erosion had been in general insignificant (e.g. Hudson, 1967), although erosion in Mediterranean regions is mentioned in documents as early as the 18th century, including the use of control measures (Chisci, 1980). Intensification of agriculture in Europe typically began in the 1950s (somewhat later in southern Europe). Many countries implemented land reallocation programmes in the 1970s: these resulted in larger fields and the adoption of agricultural techniques which exacerbated the potential for erosion (e.g. upslope tillage, autumn-planted cereal crops and heavier agricultural machinery). This led to a notable increase in erosion throughout Europe, which eventually spoke powerfully for the need for soil conservation strategies (e.g. Boardman *et al*., 2003). One result was the formation of steering committees that brought erosion and erosion modelling to the attention of the EEC DG VI (Agriculture); another was the foundation of the European Society for Soil Conservation in 1988.

Any soil conservation strategy requires some assessment of erosion rates. Erosion models provide one way of quantifying rates of erosion. Thus the first applications of erosion models in Europe were in the early 1980s, a few years after initial scientific perception of the growing erosion problem. This sequence of events is broadly similar to (but some half a century later than) the US experience. There is, however, an important difference. The first users of erosion models in Europe did not need to begin from scratch: they could make use of the knowledge gained, data collected and models devised in the USA during the preceding half century. Thus European erosion modelling owes the USA a considerable debt of gratitude for its head start.

The first wave of European model applications simply used, with little or no modification, models which had been developed across the Atlantic, particularly the USLE (e.g. Bolline, 1985; Schwertmann, 1986; Auerswald, 1988; Gabriels *et al*., 1988; Hasholt, 1988). Indeed, the unmodified USLE has continued to be used for modelling erosion in Europe to this day, often in association with GIS (e.g. Jamagne and King, 1991; Jäger, 1994; Tomas and Coutinho, 1994; Van der Knijff *et al*., 1999; 2000).

2.16.2.1.3 Subsequent Directions in European Erosion Modelling

However, by the mid-1980s a number of workers, both within the USA (e.g. McIsaac, 1990) and elsewhere (e.g. Bolline, 1985), had begun to question the extent to which the USLE could usefully be applied under conditions which differ from those areas of the USA for which the model was developed (links to the models can be found at http://soilerosion.net). Model evaluation studies began to confirm that the USLE (and models based on the USLE) might indeed perform poorly when validated against observed soil loss, particularly under European conditions (e.g. Bolline, 1985; De Roo, 1993; Favis-Mortlock, 1994, 1998b). Such poor results may be due to mis-application of the USLE, i.e. to use of the the model in ways for which it was not designed (cf. Wischmeier, 1976), such as for single events, for low-intensity rainfall, to large areal units, on stony soils or for locations in which gully erosion (which is not considered by the USLE) is the dominant form of erosion, or where the primary concern is deposition. Whatever the reason, the inherent limitations of the USLE and its direct derivatives mean that there are problems applying it in many European contexts. Nonetheless, there have been some successful applications of new European models which incorporate USLE elements

(see Table 2.16.1). This growing appreciation of the limitations of existing modelling tools generated a desire for improved models and modelling approaches which would provide reliable and defensible assessments of erosion rates for European soil conservationists, and eventually for European policy-makers.

To do this, two broad strategies were adopted, which led to two 'strands' of European erosion modelling. The first strand aims to calibrate and/or adapt existing US models in the hope that they will become more useful predictors of erosion under European conditions (e.g. ANSWERS, De Roo *et al*., 1989, 1994; CREAMS, Kallio *et al*., 1991; EPIC, Favis-Mortlock *et al*., 1991; AGNPS, Grunwald and Frede, 1999; and WEPP, Klik, 1995). Little more is said about this strand here. The aim of the second strand is to develop new models for soil conservation in Europe. This is the focus of the remainder of this chapter.

2.16.2.2 Distinctive Features of European Erosion Modelling

Given this strong US heritage of European erosion modelling, the previously mentioned distinctiveness of present-day erosion modelling in Europe is all the more remarkable. One possible reason for its individuality could be because there is a different job to do, i.e. the conditions of erosion in Europe, or because the needs of European soil conservation might differ from their equivalents in the USA. Another reason might lie in the practice of the modelling itself. Each of these is explored below.

The conditions under which soil erosion occurs throughout the countries of Europe are fully described elsewhere in this book. Therefore, we give here only a brief, very general, overview, contrasting these conditions against those of the USA.

2.16.2.2.1 Differences in Hydrology and Land Use

The rainfall which produces erosion, particularly in western Europe, is often of frontal origin. Hence it is of generally lower intensity and of longer duration than the predominantly convectional rainfall which drives soil erosion in the continental interior of the USA. Runoff generated from low-intensity rainfall may result both from saturation excess on those parts of the catchment where soils are saturated, and (once soils have crusted) from infiltration excess (Hortonian) runoff from the crusted areas of the catchment (Auzet *et al*., 1995). Variation in soil type and history (see below) will affect the propensity of different areas to crust. By contrast with the solely infiltration-excess (Hortonian) runoff, which is the norm under high-intensity rainfall of the US type, runoff-producing areas in catchments in western and northern Europe are therefore likely to possess a greater spatial variability, and this spatial variability is likely to itself vary over time (e.g. Hupert and Vanclooster, 2002). Hence erosion models which aim to represent European erosion will have to, consider either implicitly or explicitly, this spatial variation in runoff-producing areas.

There is often a greater diversity of land use within European catchments compared with their closest equivalents in the USA. Also, European field sizes are generally smaller than in the USA, particularly in western and southern Europe (however, enlargement of fields in recent decades has somewhat narrowed this difference). These factors combine to give a higher spatial heterogeneity of land use in Europe than in the USA. Given this greater patchiness in land use (in addition to the hydrological and pedological variability discussed previously), calculation of an erosion rate for a small plot-sized area, then extrapolating ('scaling up') this rate to the whole catchment is a much more dubious proposition for European catchments than it is for US catchments. For similar reasons, using a 'lumped' approach to modelling whole-catchment erosion is also not generally desirable for European conditions.

Farming techniques often differ between Europe and the USA, e.g. somewhat lighter agricultural machinery tends to be used in Europe. Hence input data for tillage operations must be recalibrated if US modelling approaches are used in Europe.

Overall, it appears that in some areas of Europe at least, soil erosion can occur under conditions of greater spatial variability than is generally the case in the USA. If this is true, then successful modelling of European erosion may be expected to require a stronger emphasis on the effects of spatial patterning compared with the USA.

2.16.2.3 Social Influences on the Practice of Soil Erosion Modelling

Science is a social activity, although one with certain novel features, and like all social activities it has tended to develop its own cultures and traditions (see, e.g., Harvey, 1969; Livingstone, 2003). The branch of science known as erosion modelling is a relative newcomer, being (at most) only about 70 years old. Already, though, it has developed something of its own outlook, complete with its own customs, mores and taboos, and this outlook is not spatially homogeneous. For example, the disciplinary 'homes' of erosion research are, in general, agricultural engineering and hydrology in the USA and Canada and geomorphology/physical geography (together with other disciplines, such as civil engineers and agronomists) in specific parts of Europe. Therefore, although the policy drivers for soil erosion modelling have been broadly similar on both sides of the Atlantic, the scientific work of erosion modelling in Europe and North America has been carried out by groups of scientists separated not just by the Atlantic Ocean but also by their own disciplinary traditions. As a consequence, they have had slightly different aims (Anderson and Sambles, 1988; Parsons and Abrahams, 1992; Favis-Mortlock *et al*., 2001). Geomorphologists view erosion as one of the processes which operate upon the landscape, and use erosion models to improve their understanding of the way in which this occurs. Agricultural engineers and hydrologists view erosion more pragmatically, in terms of its effects upon crop productivity or water quality, for example.

There are a number of outcomes from this cultural divide in erosion modelling. An important one is that applications of strongly empirically based models, such as the USLE, are often viewed rather differently on different sides of the Atlantic. Empirical factors, such as the so-called 'factor of safety', are an everyday fact of life for the practising engineer. However, since the so-called Quantitative Revolution of the 1960s (Harvey, 1969), the geomorphologist shares with the physicist a desire to reduce, at least in principle, all earth surface processes to their underlying physical fundamentals. In such a quest, empirical factors and other black-box approaches are best avoided.

More speculatively, the predominantly geographical or geomorphological background of European erosion modellers may (in addition to the physical factors discussed above) contribute towards a somewhat greater European emphasis on catchment-scale models rather than plot- or field-scale models, and also a rather stronger emphasis on the topographical and pedological aspects of catchments. Speculation aside, however, one clear outcome of the differing histories of erosion modelling in Europe and the USA is that there was a much greater diversity of potential modelling approaches/tools available to the first European modellers, owing to the later start in Europe. For example, considerable effort was put into the nomograms of the USLE: these ingeniously create on paper what is, in modern terms, a multi-factorial database lookup. Such technology enabled US field conservationists to compute USLE estimates in the field with only paper and pen (and perhaps a slide rule). More recent modelling efforts, including those of European modellers, could assume a near-universal access to powerful computers, even in the field. There is, however, some indication that this easily available computer power can lead to 'bloat' in computer-based erosion models: in other words, models which make strong demands upon computer power, not for the purposes of simulating erosion, but for such things as elaborate error-checking user interfaces, or graphics which might be more efficiently handled by a purpose-built graphing package (Favis-Mortlock *et al*., 2001).

In addition, the considerable effort put into the development of the USLE (and USLE-based models) by the US community of erosion researchers and the accumulation of a considerable body of USLE-related expertise have meant that there is an understandable unwillingness to move away from such approaches. This is a

consequence of the 'erosion model life cycle' (Favis-Mortlock *et al*., 2001). By contrast, European modellers have shown no such compunctions.

2.16.2.4 The Policy Context of European Erosion Modelling

The relatively recent inclusion of the impacts of (accelerated) soil erosion in European laws and policies means that only recently has there been a need for erosion models to supply policy advice. At first, results from the 'first generation' of European applications of US models were often used to support policy. This tendency has been exacerbated by the fact that home-grown European erosion models have always been more research tools than 'agricultural extension' or soil conservation tools: this is because they have (without exception?) been created by academic institutes rather than governmental agencies. Why is this?

In the context of erosion modelling, there has always been a close cooperation between research institutes and local government agencies in Europe in such activities as obtaining data and providing experimental catchments or test fields on experimental farms. However, each country in Europe has its own environmental and agricultural policies, which are based on European laws and guidelines, often 'translated' to the national situation. This means that these local government agencies vary greatly from country to country: the research components of such agencies vary from agricultural research institutes, to agencies in charge of surface water, the environment, to more or less well-organised farmers associations, etc. There is therefore a great variety in players in the field, each with their own background and their own emphasis on erosion problems. Thus the academic developers of erosion models in different European countries have all had very different opportunities for collaboration with their government-funded colleagues.

The development of European erosion models has in some cases been funded by national or European research programmes, but such instances have (without exception?) been a one-time funding for the development of the model only. After that, the maintenance of the model has, in many cases, fallen to the hands of the individuals who created them, sometimes supported by their research institutes, sometimes not. This situation has tended to create a constant turnover: old models disappear, new models are created. In such a spatially and temporally fragmented situation, discussion and exchange of model data are not encouraged; neither is source code likely to be shared between the groups; neither is the situation likely to promote the establishment of a concerted Europe-wide network of data acquisition, even something as basic as a network of standardised erosion plots.

Noneless, in spite of a lack of pan-European organisations (cf. USDA) for modelling collaboration, there have been some groupings for parts of Europe which have (among other roles) encouraged collaboration in modelling with some success, e.g. MEDALUS for Mediterranean Europe. Also the IGBP-GCTE SEN, although global in focus, has had a strong European component, e.g. via its model comparison exercises (Favis-Mortlock, 1998a; Jetten *et al*., 1999). Erosion modelling has also benefited from the European COST programme, with the now-defunct COST Action 623 ('Soil Erosion and Global Change') and the current Action 634 ('On- and Off-site Effects of Soil Erosion and Runoff'). These have been and are long-overdue platforms where much useful exchange has taken place between model developers and user groups.

2.16.3 EROSION MODELS WHICH HAVE BEEN DEVELOPED IN EUROPE

Table 2.16.1 gives an overview of the several currently used European erosion models. Some clarification is first needed to explain our choice of models. The models included in this chapter are those which are actively maintained, easily available to a wider public and regularly used by one or more research or user groups. To

facilitate the description, the models are classified according to spatial scale: starting at the plot level (class 1) and going up via hillslope (class 2), small catchments (class 3) and large-scale catchments or river basins (class 4) to a national or European scale (class 5). Note too that the spatial and temporal scales mentioned in the overview represent the average scales on which the models have been applied, not the scale on which the authors claim they operate! Models often can be applied outside the range shown here, although the modelling results may not be optimal.

An overview of these models and their main characteristics can be found in Table 2.16.1. We fully realise that the model descriptions are not exhaustive and we encourage readers to find more complete information from publications, and/or on the Internet (links to the models can be found on http://soilerosion.net). The emphasis in this table is on the models' representations of erosion, transport and deposition processes, more than on hydrological processes. It is also important to note that if two models have the same basis for a process, they still may not necessarily apply these principles in the same way. If this is the case, these models may still behave entirely differently compared with each other.

Also not included in Table 2.16.1 are models that have components which represent the generation and transport of sediment by overland flow, but which simulate only water, such as MIKE-SHE (which uses MIKE11 only for sediment transport in the larger channels; Refsgaard and Storm, 1995), or TOPMODEL (Beven, 1997), or focus mainly on groundwater movement and river discharge, such as TOPKAPI (Ciarapica and Todini, 2002).

2.16.3.1 Input Data

In terms of data input, the smaller scale models all require maps of parameters that are directly used in model algorithms: none of the models use built-in conversion tables or transfer functions to translate, for instance, land-use types, soil types or soil classes to the parameters required in the model, except for the expert model STREAM (see below). This reflects perhaps the fact that soil data in Europe is irregular and each country uses its own soil classification system, so that pragmatically it makes more sense to use measurable parameters directly, rather than to derive them from soil data. The same remark can be made about cultivation practices that differ from country to country. Plant- or crop-related processes are somewhat under-represented in the models, e.g. not all models consider rainfall interception by the canopy and related processes (leaf drip, stem flow). Almost all models, however, include plant cover in their splash detachment modules, and many models present guidelines in their manuals to increase, for instance, flow resistance or soil strength based on plant cover/density (see, e.g., the EUROSEM manual; Morgan *et al*., 1998). Many discussions within the erosion community (made possible in part by the COST 623 erosion network) have led to the integration of the effects of farm management and tillage practices in models at field-scale and larger. Such discussions have led to an increased awareness of the relation between crop rotation cycles and farm management, and to the periodic (often seasonal) changes in soil structure leading to surface sealing and changes in surface roughness (see, e.g., Le Bissonnais *et al*., 1995; Fohrer *et al*., 1999; Kamphorst *et al*., 2000). Also, the relationship between runoff connectivity in a catchment and tillage practices within the catchment has received a great deal of attention (see, e.g., Souchère *et al*., 1998; Takken *et al*., 2001). These agricultural factors are increasingly incorporated into European erosion models; for example, STREAM (Cerdan *et al*., 2002), LISEM (Jetten and De Roo, 2001) and WATEM/SEDEM (Van Oost *et al*., 2000; Van Rompaey *et al*., 2001) all have special GIS algorithms to deal with agriculturally induced flow direction.

A source of data that is not often specifically mentioned or used is remotely sensed data. Only the largest scale models (SEMMED and PESERA) specifically use databases derived from remote sensing as part of the model structure. Some models may derive plant-related variables (cover LAI) from remotely sensed data, but this source of data is not yet well integrated into modelling procedures.

TABLE 2.16.1 European erosion models and a summary of the main processes and characteristics

	RillGrow	MMF	PSEM-2D	Erosion2D/3D	LISEM	Eurosem	SMODERP	MEFIDIS
Scale	Plot	Plot (point)	Hillsope $<0.01\ km^2$	Hillsope $<0.01\ km^2$ catchment $<1010\ km^2$	Catchment $<10\ km^2$	Catchment $<10\ km^2$	Catchment $<10\ km^2$	Catchment $<50\ km^2$
dt (typical)	1×10^{-6} s	1 year	s	min	s	s/min	s	s
dx (typical)	<0.01 m	n.a.	1–10 m	5–100 m	5–100 m	slope + channel elements	1 m	5–100 m
GIS	Raster	n.a.	Raster	Raster	Raster	Polygon	n.a.	Raster
Rainfall	Event	Annual	Event	Event	Event	Event	Event	Event
Interception	No	Fraction of P	no	Not specified	Canopy storage, cover, LAI	Canopy storage, cover, LAI	Canopy storage, cover	Canopy storage, cover, LAI
Infiltration	Subtraction infiltration rate	Soil storage overflow	?	Green and Ampt	Green and Ampt or Richards/Darcy	Smith and Parlange	Phillip	Green and Ampt
Surface storage	Yes	No	?	No	Random Roughness, slope	Tortuosity, slope	Roughness	Depression storage
Runoff	Yes, similar to Manning	Discharge is infiltration surplus	Darcy–Weisbach	Darcy–Weisbach, flow based on momentum	Manning	Manning	Manning	Manning
Routing	Decision-based algorithm	n.a.	Full Saint-Venant	Kinematic wave	Kinematic wave	Kinematic wave	Kinematic wave	Kinematic wave
Network	Dynamic, based on water level and local slope	n.a.	Topographic	Topographic, generated by the model from DEM	Topographic/ tillage	Hillslope/ channel elements	Profile	Toporaphic
Suspended matter	Single texture class only	n.a.	D50	9 texture classes	D50, or 5 texture classes	D50	Only total sediment	D50
Splash detachment	kinetic energy	Kinetic energy, splash erodibility	Intensity, turbulence factor, erodibility	Momentum of rainfal	Kin energy, aggr stab	Kinetic energy, inter-rill erodibility	1 equation for splash and flow detachment	
Transport (and transport capacity)	Shear stress	Slope-discharge function	Shear stress	Shear stress	Unit stream power	Unit stream power	Shear stress	Shear stress

Flow detachment	Based on occurrence of turbulent bursts	Slope-discharge function, erodibility	Erodibility, TC limited	Erodibility, TC limited	Cohesion, TC limited	Cohesion, TC limited	1 equation for splash and flow detachment TC surplus	Erodibility, TC limited
Deposition	Sediment surplus	Implicit, sediment surplus	Settling velocity, TC surplus	Settling velocity, TC surplus	Settling velocity, TC surplus	Settling velocity, TC surplus	TC surplus	Settling velocity, TC surplus
Type of erosion	Splash + rill	Not specified	Splash + rill	Not specified	Splash + rill + gully (separate module)	Splash + sheet + rill	Not specified	Not specified
Additional features, remarks	Micro-scale model, based on self-organising systems		A 3D raster is used to represent the hillslope	Two versions: a hillslope and catchment verison; momentum based algortihms	Additional modules: multiple sediment classes, nutrient loss and gully erosion	Additional modules: gully erosion	Modules of soil erosion (critical slope lenght) and surface runoff (runoff hydrographs, soil loss)	
Internet (www.)	http://soilerosion.net/rillgrow/2/rg_2.html	No	No	http://www.geog.fu-berlin.de/~erosion/	http://www.geog.uu.nl/lisem	http://www.silsoe.cranfiled.ac.uk/nsri/research/erosion/eurosem.htm	http://departments.fsv.cvut.cz/k143/	http://gasa.dcea.fct.unl.pt/mefidis/
Scale	Catchment $<10\ km^2$	Catchment $<0.01\ km^2$ $5000\ km^2$	Catchment $10–1000\ km^2$	Catchment $100–50\,000\ km^2$	Catcment $100–30\,000\ km^2$	Catchment $100–10\,000\ km^2$	Catchment $100–10\,000\ km^2$	$>10000\ km^2$
dt (typical)	Lumped	1 year	15 min–2 h	1 year	1 day	1 month	1 year	1 month
dx (typical)		1–50 m	10–1000 m		100–1000 m	3 km	n.a.	1 km
GIS	Polygon/raster	Raster	Raster	Raster	Polygon	Raster	n.a.	Raster
Rainfall	event	Annual	Events, continuous	Annual	Daily	Monthly	Annual	monthly + intensity

(*Continued*)

TABLE 2.16.1 *(Continued)*

	STREAM	WATEM/SEDEM	SHETRAN	SEMMED	SWIM	RHINEFLOW	FSM	PESERA
Interception	Tables based on vegetation	no	Canopy storage, cover, LAI	Fraction of P	Canopy storage, cover LAI	no	n.a.	no
Infiltration	Tables based on crusting class	no	Richards/Darcy	Soil storage overflow	Green and Ampt	Soil storage overflow	n.a.	Soil storage overflow
Surface storage	Tables based on crusting class	no	no	no	no	no	n.a.	no
Runoff	n.a.	n.a.	Manning	Discharge is infiltration surplus	modified Curve number method	Discharge is infiltration surplus	n.a.	Discharge is infiltration surplus, flow length limited by relief factor
Routing	Spatial accumulation of net runoff	Based on LS algorithms	Full Saint Venant	spatial accumulation of net runoff	Muskingam routing scheme	Spatial accumulation of net runoff topographic		n.a.
Network	Topographic+ tillage combined	Topographic/ tillage	Topographic	Topographic	Topographic	Topographic	Slope and basin shape determine runoff factor	Topographic, special algorithm for local slope
Suspended matter	D50	n.a.	multiclass	n.a.	n.a.	n.a.	n.a.	n.a.
Splash detachment	n.a.	Kinetic energy, erodibility	squared rainfall momentum, splash erodibility	kinetic energy, splash erodibility	kinetic energy, EI30, erodibility	kinetic energy, EI30, erodibility	Vegetation cover factor	Rainfall power function splash erodibility
Transport (& Tr. Capacity)	n.a.	LS routing algortihm based	shear stress	slope-discharge function	n.a.	n.a.	Lithology-derived sediment source factor	Slope-discharge function, erodibility, vegetation
Flow detachment	n.a.	Erodibility, RUSLE based	erodibility, TC limited	no	erodibility, MUSLE based	erodibility, RUSLE based	n.a.	Slope-discharge relationship
Deposition	n.a.	Implicit	Settling velocity, TC surplus	Implicit, sediment surplus	no	Implicit ?	n.a.	no
Type of erosion	Rill + gully	Splash + rill	Not specified	not specified	Not specified	Not specified	gully	Not specified

Additional features, remarks	Dynamic version using kinematic wave under development	Additional modules: tillage erosion using diffusion, ponds and reservoirs included	A landslide component is under development	Based on an older version of MMF: no flow erosion	Focus on nutrient transport, based on SWAT, MUSLE	Snowmelt module included. Rhineflow is primarily used for climate chage analysis	Based on a non-linear combination of user assigned factor scores	Frozen soil module included. Vegetation growth is modelled, influences erodibility and cover parameters.
www	http://www.kuleuven.ac.be/geography/frg/leg/modelling/watem/index.htm	http://www.ncl.ac.uk/wrgi/wrsrl/rbms/rbms.htm#SHETRAN	http://www.frw.ruu.nl/fg/demon.html	http://www.brc.tamrns.edu/swat/othermod/swim-des 5p.htm	http://rhine.geog.uu.nl/rhineflow.html	no	http://pesera.jrc.it/	

2.16.3.2 The Models Ordered by Scale

2.16.3.2.1 Class 1: Plot Size Models

On the smallest scale (several square metres), the Rillgrow model (Favis-Mortlock *et al*., 2000) simulates the generation of rills on a DEM of soil surface roughness. The model is based on the concept of self-organisation. All relationships in the model are applied 'locally' (i.e. between a single cell of the DEM and its neighbours), while the model produce an emergent 'global' response: a rill network on the whole DEM. Flow is stochastically routed from cell to cell of the DEM, based on the energy slope of the water's surface. This flow then modifies the DEM by means of erosion or deposition. This in turn modifies the routing of subsequent flow; eventually, rills and rill networks are formed. Flow erosion is calculated using a stream power approach based on the notion of turbulent bursts (Nearing, 1991). A diffusive splash erosion submodel (Planchon *et al*., 2000) is loosely coupled to the flow algorithms; however, simulation of infiltration is very crude. The model has been tested by comparing the spatial patterns of the simulated rills with real rills on laboratory flumes of field plots. Computational requirements considerably constrain RillGrow's usage on areas larher than a small plot.

More classical in concept is the Morgan, Morgan and Finney model (Morgan, 2001), which is a good example of one of the few lumped models created in Europe. It is not specifically made for Europe as the parameter database includes crop and soil factors from tropical countries. So far it has been applied mainly in Africa. It calculates average annual erosion, deposition and soil loss for a single spatial unit. In contrast to the USLE, all major hydrological and erosion processes are represented: interception, infiltration, discharge, splash detachment based on rainfall kinetic energy and flow detachment and transport capacity based on discharge and slope.

2.16.3.2.2 Class 2: Hillslope Models

PSEM-3D (still under development by the Laboratoire d'études de Transferts en Hydrologie et Environnement, Grenoble, France) and Erosion 2D (Schmidt and Manersberger, 2004) are specifically hillslope models, although all catchment-based models in class 3 can operate at a hillslope scale, simply by reducing the number of spatial elements. Both PSEM-3D and Erosion 2D are grid-based models based on the same general physical principles as the class 3 models, described in the next section. Specific to Erosion 2D and Erosion 3D (the catchment version; see Schmidt *et al*., 1999) are the use of momentum equations for both splash and flow detachment. This momentum-based approach has necessitated the development of a large model-specific database.

2.16.3.2.3 Class 3: Small Catchment Models

This class encompasses mostly event-based models that operate on small catchment scales ($<10\,km^2$, usually simulating catchments of the order of 0.01–0.5 km^2). These models use breakpoint rainfall data, a single- or multi-layer Green and Ampt infiltration (or a comparable method) and a kinematic wave procedure combined with a Manning or Darcy–Weisbach friction factor to route the water over a network. A notable difference in hydrology can be found in the simulation of surface storage, as it is sometimes not included at all, based on empirical functions of various roughness indices or directly based on a surface storage parameter. A good example in this class is EUROSEM (Morgan *et al*., 1998) that is developed as a European effort involving several research teams from different countries. Using the kinetic energy of rainfall and unit stream power to generate sediment, the model is based on the principle of transport capacity deficit (see, e.g., Foster, 1982) whereby it is assumed that the sediment concentration in overland flow will tend towards the maximum transport capacity. Flow detachment occurs when this transport capacity is not reached and is mitigated by a soil strength parameter. Deposition occurs when there is a surplus of sediment. In spite of this generally similar approach, there are clear differences between the models: splash detachment may be based on different driving

forces (rainfall intensity, kinetic energy or momentum) and flow detachment/transport capacity may be based on a form of stream power or shear stress of the flowing water. Also, different soil strength parameters are used, varying from cohesion or aggregate stability to a more general 'soil erodibilty' (note that all models use different strength parameters for splash and flow detachment). All models seem to simulate essentially rill erosion in view of the parameter range used, although sometimes it is unspecified. Some models have special gully erosion modules (indicated in Table 2.16.1).

There are two models on this scale that are distributed in space but lumped in time: STREAM (Cerdan *et al*., 2002) and WATEM/SEDEM (Van Oost *et al*., 2000, Van Rompaey *et al*., 2001). The STREAM model can be considered a rule-based expert system. It is designed for areas that have a clear surface sealing process, where rainfall causes a loose soil after tillage to develop gradually a crust in the course of the season. Observed crusting classes form the input of STREAM, which then uses a set of tables for surface storage and infiltration rates and combines these with plant cover to calculate runoff and erosion. Excess rainfall and sediment are accumulated once per event over a combined topographic/tillage network. WATEM/SEDEM simulates annual erosion based on the RUSLE, but using a 2D algorithm to replace the upslope length by the unit contributing area, including convergence and divergence of flow in the landscape based on the aspect of the slope (Desmet and Govers, 1996). The erosion rate is calculated using a local transport capacity as upper limit. Originally WATEM and SEDEM were two models that have been merged. WATEM operates until now on smaller scales ($<1\ km^2$) and is the only model that simulates tillage displacement using a single parameter-based diffusion-type equation (Govers *et al*., 1994). SEDEM has been applied on large scales ($>1000\ km^2$) and is therefore also a 'class 4' model. It is important to note that it has been calibrated and validated on that scale against sediment accumulated in retention ponds (Van Rompaey *et al*., 2001).

2.16.3.2.4 Class 4: Large-scale Models

On a large catchment to river basin scale (100–100 000 km^2), the models included here both simulate continuously in time and are therefore based on a complete water balance, including evapotranspiration and groundwater flow, or are lumped in time to provide monthly or annual simulations and then use simpler process descriptions. Land use is represented at various levels of complexity: from crop growth models to a user-defined input, sometimes derived from satellite images. Below are four examples of process-based models in order of increasing spatial scale and decreasing physical complexity.

SHETRAN (Ewen *et al*., 2000) is one of the more comprehensive, physically based models in this scale class. It is specifically designed to model transport of chemicals through various pathways (including sediment related), on a continuous basis. Vertical soil water and lateral ground water movement is physically based, and runoff routing is done with the Saint-Venant equations. SHETRAN is the only model that has a component to simulate shallow landslides, triggered by groundwater fluctuations.

Less complex is the SWIM model (Krysanova *et al*., 1998), which uses small hydrological sub-catchments as the basic spatial unit that are linked with a Muskingam flow routing scheme. The model principles are largely based on SWAT: overland flow is generated using a modified curve number method, while it also simulates interflow and groundwater flow. Sediment generation is based on the modified USLE, which has flow detachment based on a combination of discharge and LS factor and splash erosion based on the rainfall kinetic energy and intensity. SWIM is primarily a model for research of nutrient dynamics.

SEMMED (De Jong *et al*., 1999) has the same process descriptions as the MMF model (see above) applied to a much larger scale. Vegetation-related input data are specifically derived from satellite images. SEMMED simulates only a runoff component which is lumped in time and distributed in space, i.e. a transport capacity limited accumulation of runoff and sediment over a topography-based network. So far it has been used for an assessment of annual erosion in Mediterranean areas. RHINEFLOW (Asselman *et al*., 2003) is similar in structure to SEMMED in that it accumulates net flow over a topography network. It recognises a fast flow

component and base flow component. Its erosion part is based on a German version of the RUSLE, combined with a sediment delivery ratio derived from the slope, a distance factor and a runoff coefficient. Its primary use so far is for long-term climate change studies and studies of the dynamics of chemicals. RHINEFLOW has been calibrated against discharge and sediment loads of the river Rhine on a monthly time-step basis.

Finally in this scale class we mention the 'factorial scoring model' (FSM) (Verstraeten *et al.*, 2003), which predicts the specific sediment yield of a catchment based on a nonlinear equation involving the catchment area and five weighted additional factors: topography, vegetation, gullies, lithology and slope. Although it is nonspatial and only predicts annual sediment yield, FSM has the advantage that it is simple and it has been used with good results in Spain and Italy (De Vente and Poesen, 2005).

2.16.3.2.5 Class 5: European-scale Models

On this scale, mostly erosion risk estimates have been made applying USLE-type concepts on European-scale datasets (Van der Knijff *et al.*, 2000). A good overview of developments on this scale is given by Grimm *et al.* (2002). Le Bissonnais and Daroussain (2001) made a European erosion assessment which has become known as the 'INRA approach' (not included in Table 2.16.1). This method focuses on pedo-transfer functions that generate soil crusting susceptibility and erodibility classes, combined with the CORINE land cover database, relief factors from a 1-km grid cell DEM and multi-annual rainfall data. This effort has led to the development of the PESERA model, which attempts to be 'physically meaningful at this scale'. PESERA estimates average discharge in a 1-km grid cell from infiltration surplus and combines this with a DEM-derived relief factor to simulate both flow detachment and sediment transport. These flow processes are modified by land cover and soil erodibility. Splash detachment is related to a power function of rainfall depth and splash erodibility. The PESERA model is still under development and currently produces monthly European erosion maps, but it is currently being tested for much smaller areas in the north-west France.

2.16.3.3 Models: Summary Remarks

A few interesting observations can be made when comparing the models listed in Table 2.16.1:

- All models have been calibrated and validated at all scales for which they are constructed (even the largest scale).
- All spatial and temporal scales exist: from micro plot to European scale and from event-based with time steps of less than 1 min to continuous simulation models with hourly or monthly time steps. However, not all combinations of spatial and temporal scales exist: the small catchment scale models are almost all event based (apart from WATEM/SEDEM, which is lumped) and therefore do not include processes such as evapotranspiration and groundwater. The continuous models that do incorporate these processes are all on a larger scale. This may limit the type of analysis that can be done on this scale, regarding, for instance, the effect of land use or climate change.
- All models, regardless of scale, are based on some form of transport capacity which may limit the potential erosion. The transport factor seems to be based on one of three concepts: a power function combining slope or LS factor with discharge, the potential shear strength or the stream power (which are, of course, physically related). All models recognise splash (inter-rill) and flow detachment and use separate erodibility parameters for these processes.
- Most models simulate essentially rill erosion, although of course on larger scales the type of erosion it unspecified. However, other forms of erosion are also simulated: WATEM includes tillage displacement, some models have separate sheet erosion equations (not elaborated here), LISEM and EUROSEM have a gully incision module (while others report good results in simulating gully erosion) and SHETRAN has a module to simulate shallow land slides and slope instability.

- Very few models use remote sensing input (only SEMMED and PESERA specifically use it). However, the level of detail of remote sensing data available has dramatically improved with multi-spectral data, very high-resolution data, automatic DEM extraction, etc. A reacquaintance between modellers and remote sensing specialists may be fruitful.

2.16.3.4 Model Performance

One of the main questions addressed by the discussions generated by platforms such as COST623 and IGBP-GCTE has been the model performance, especially for the prediction of off-site effects. We must first stress that for all models in all classes proof is given of medium to good performance in predicting total discharge and total soil loss for individual cases, certainly after calibration for a specific set of circumstances. Several models have also been tested with ^{137}Cs data (e.g. De Roo and Walling, 1994; Owens and Walling, 1998; Porto *et al*., 2003). Several studies have been carried out in which models from classes 2 and 3 have been compared with each other for predictive quality, both in an orchestrated way (Favis-Mortlock, 1998b; Jetten *et al*., 1999) and in individual model comparisons (e.g. De Roo, 1993; Vlnasova *et al*., 1998). The results of these studies can be summarised as follows. Although the models can be calibrated well to individual situations (runoff events), the 'unsupervised' prediction of future situations has to be treated with great caution. Generally different sets of calibration parameters are needed for events of small and large size and for different seasons in the case of agricultural land use. The performance is best for lumped variables such as total runoff and total soil loss, less well for hydrographs and sedigraphs. When comparing the performances for these lumped results (e.g. annual soil loss), the more empirical models that are lumped over time (so-called USLE derivatives) perform just as well as physically based models that attempt to simulate the hydrological and erosion processes in detail. Considering also that lumped models need less input data, they can be more attractive to a decision maker.

One of the reasons for the mediocre performance of physically based models is that the total erosion and deposition in a catchment is usually far greater than the net soil loss, which therefore has a large uncertainty. Moreover, the spatial distribution of contributing areas is often unknown and can be completely misinterpreted (Jetten *et al*., 1996), while the results of the STREAM model show the strength of using qualitative spatial knowledge of potential contributing areas. Physically based models, however, produce a lot more information about the processes and are therefore more attractive for research purposes. Both types have their merits and comparisons are actually fairly difficult, mainly because of the different temporal resolutions.

Some studies have evaluated the quality of the predicted spatial patterns. Erosion patterns appear to be, in part, chaotic in nature (in that they can be very sensitive to tiny variations in initial conditions, at some locations within the catchment). Possibly as a result, although almost all current models seem capable of producing credible patterns that 'look' good, usually they are not capable of producing the observed spatial erosion and deposition at the exact location (Jetten *et al*., 2003). However, also on a lower resolution, comparing the simulated erosion of a catchment aggregated per field with total soil loss measured form those same fields for the same event met with mediocre success (Takken *et al*., 1999, 2005). Some progress in predicting and modelling gully locations has been made by introducing observed landscape factors (Jetten *et al*., in press). Several researchers, however, have indicated that it may not be possible to improve on these results, in part because of the apparently chaotic components of flow routing and erosion and also because of the large uncertainties involved in the predictions (Favis-Mortlock, 1994; Quinton, 1997), and also because of the notion of model equifinality (e.g. Beven, 1996). The larger the scale on which the model operates, the more difficult it is to evaluate its performance. The class 4 models have been calibrated and tested, for instance SHETRAN with high resolution river data in the UK and Mediterranean (Parkin *et al*., 1996; Lukey *et al*., 2000), RHINEFLOW with the monthly discharge and sediment load of the river Rhine

(Asselman *et al*., 2003) and FSM on annual sediment yield of a series of catchments in Spain (De Vente *et al*., 2005).

2.16.4 PRESENT AND FUTURE

In this chapter, our aim has been to review currently used European erosion models. Whereas a review of *all* such models is, of course, the most desirable outcome, constraints on time and length mean that we have been forced to limit ourselves to those models that have a published track record. If this approach disappoints or offends anyone, then we apologise here.

The development of soil erosion models in Europe is now well established. Models of European origin are now used both inside and outside Europe. In terms of spatio-temporal focus, there is some overlap but little redundancy in the models: models have been created for every spatial and temporal scale. This does not mean, however, that all combinations of time and space are considered by European models: on the hillslope and catchment scales the models tend not to be capable of temporally continuous simulation, but instead usually can do only temporally lumped simulations (e.g. average annual erosion) or single event-based simulations of hydrographs, sedigraphs and erosion patterns. The lack of continuous simulations at this spatial scale could limit the type of analysis possible (e.g. climate change).

There are both similarities and differences in model conceptualisations and structure, e.g. in process descriptions and in operational conditions (such as GIS used, type of input data required and type of output data produced). Also, it is important to note that all models listed here, regardless of scale, have been calibrated and validated in one way or another. In spite of the wide range of spatial and temporal scales in modelling techniques, it is recognised that the data to drive these models do not exist or are scarce. The more processes a model encompasses, the more parameters need to be specified and measured. Several activities are currently taking place in Europe (e.g. within the COST634 network, the successor to COST623) that aim at gathering and collating the runoff and erosion databases that exist in the member countries. It will be interesting to see how this essential exercise changes our views regarding the matching of available data and model requirements, and thus influences directions of research for the future.

There is still much improvement possible in European soil erosion models. For instance, most models assume rather classical process descriptions, in which many soil and surface parameters are assumed constant in time. However, some of the most important parameters such as hydraulic resistance (Takken and Govers, 2000) or saturated hydraulic conductivity (e.g. Wainwright, 1996) are not constant in time but may change in the course of a rainfall event. Recent research has shown that soil structure changes caused by tillage operations, in combination with rainfall impact and crop growth, provide conditions that cause a much more dynamic environment at the soil's surface and just below than the models (and modellers) currently assume (e.g. Fohrer *et al*., 1999).

Improvements can also be made in the spatial aspect of models: lack of knowledge on the spatial distribution of sources and sinks in a catchment often results in calibration of model parameters which assume a much too homogeneous pattern of runoff and sediment dynamics. The result may be, e.g., a correct hydrograph for the wrong reasons (Favis-Mortlock *et al*., 2001). Validation of simulated spatial patterns is still hardly ever attempted, although from discussions at COST meetings a good understanding of sources and sinks in a catchment is seen as a major improvement and a way of overcoming the problems of equifinality. Possibly remote sensing can play an important role here, as the resolution of remotely sensed images has greatly improved since then (e.g. Quickbird and Ikonos have a resolution of <1 m).

The focus in modelling and model development is currently shifting from the 'classical' on-site effects within a catchment to the estimation of off-site effects of erosion, such as the impact of chemicals in runoff and of suspended sediment in the downstream environment. In parallel with this, the purpose of European erosion modelling is apparently shifting from the ability to produce a correct hydrograph and/or soil loss

amounts to, e.g., applying a model to define erosion risk classes that may be used to determine agricultural subsidies or penalties. Possibly this widening of modelling purpose also widens the range of acceptable predictive quality (since for defining erosion risk classes, order-of-magnitude estimates are likely to be good enough); or on the contrary this may put a greater strain on the perceived ability of European erosion models to 'correctly' estimate erosion (since money is increasingly involved).

Perhaps the most encouraging indication for the future of European erosion modelling is that, in our experience, model developers are very aware of the limitations of their models, maybe more so than are model users. Continued discussion and exchange between all those who are involved in building and using European erosion models is therefore imperative; networks such as COST623 and COST634 have a major role to play here.

2.16.5 ACRONYMS USED

AGNPS	Agricultural Non-point Source pollution model
ANSWERS	Areal Non-point Source Watershed Environment Response Simulation
CORINE	Coordination of Information of the Environment
CREAMS	Chemicals, Runoff, Erosion and Agricultural Management Systems
DEM	Digital Elevation Model
EEC-DG	European Economic Community – Directorate General
EPIC	Erosion–Productivity Impact Calculator
EUROSEM	European Soil Erosion Model
FSM	Factorial Scoring Model
GIS	Geographical Information System
GLEAMS	Groundwater Loading Effects of Agricultural Management Systems
GUEST	Griffith University Erosion Erosion System Template
KINEROS	KINEmatic Runof and erOSion model.
LISEM	Limburg Soil Erosion Model
MEFIDIS	Modelo de Erosão Físico e DIStribuído
MIKE-SHE	SHE stands for Système Hydrologique Européen
MMF	Morgan–Morgan–Finney model
MUSLE	Modified Universal Soil Loss Equation
PESERA	Pan European Soil Erosion Risk Assessment
PSEM-3D	Physical Soil Erosion Model
RUSLE	Revised Universal Soil Loss Equation
SEDEM	Sediment Delivery Model
SEMMED	Soil Erosion Model for the Mediterranean
SHETRAN	Système Hydrologique Européen TRANsport
SLEMSA	Soil Loss Estimator for Southern Africa
SMODERP	Simulační Model povrchového odtoku a Erozních Procesů
STREAM	Sealing Transfer Runoff Erosion Agricultural Modification
SWAT	Soil and Water Assessment Tool
SWIM	Soil and Water Integrated Mode
SWRRB	Simulator for Water Resources in Rural Basins
TOPKAPI	TOPographic and Kinematic Approximation and Integration
TOPMODEL	Topographic Model (?)
USLE	Universal Soil Loss Equation
WATEM	WAter and Tillage Erosion Model
WEPP	Water Erosion Prediction Project

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2.17

Existing Soil Erosion Data Sets

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2.17.1 INTRODUCTION

Geomorphological evidence from all over the world clearly shows that human alteration of the natural vegetation cover – as a prerequisite for agricultural activities (among others) – causes accelerated soil loss. In Europe, written recognition of this inadvertent consequence of human action and its impacts on the environment dates back to early Greek scholars such as Sophocles and Plato. 'To-ing, froing, with the plough-teams going, Tearing the soil of her, year by year' – these two lines from Sophocles' *Antigone* (cited in Glacken, 1967, p. 120) is probably the first reference to what nowadays is called tillage erosion. Furthermore, geoarchaeological evidence and written accounts prove that measures against soil erosion have been an issue (for farmers and others) throughout the agricultural history of Europe. Strongly eroded abandoned field sites, currently either under forest cover as in some sites in the northern loess zone (Bork, 1985; Bork *et al.*, 1998) or already bare of soil as in some places in the Mediterranean (van Lynden, 1995, p. 13), highlight the ultimate consequences of soil erosion in Europe, where 90 % 'of the long-settled lands . . . are degraded to some degree' (Bot *et al.*, 2000, p. 23).

Despite all the early evidence for the importance of soil erosion in Europe and intense scientific reviews of the problem since the 1950s, including the first nation-wide soil erosion assessments (e.g. Flegel, 1958; Richter, 1965), it took nearly another four decades until the first European-wide assessment of the current state of soil erosion was presented (de Ploey, 1989). Together with other international activities (Oldeman *et al.*, 1991; CEC, 1992; RIVM, 1992; van Lynden, 1995), this fostered the European-wide integration of soil erosion research and the collection of harmonised continental-scale data sets to be used in GIS analysis (e.g. Heymann *et al.*, 1994; King *et al.*, 1994). Finally, the past few years saw the development of two standardised approaches

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to evaluate rainfall-driven soil erosion across large parts of Europe (van der Knijff *et al.*, 2000; Kirkby *et al.*, 2004). These almost EU-wide standardised approaches are highly desirable as they provide comparable information on a considerable portion of the soil erosion problem in the area considered. However, problems inherent to the spatial scale (1 km^2 resolution) of the input data sets used in these approaches should not be overlooked (van der Knijff *et al.*, 2000, pp. 23*ff.*; Kirkby *et al.*, 2004, pp. 8*ff.*). In addition, both data sets are restricted to soil erosion due to rainfall and therefore do not cover northern Europe, where soil erosion due to snow melt is important (cf. Chapter 1.1).

The scope of this chapter is to give an overview of potential sources of information on soil erosion and to summarise and review available national data sets on soil erosion in Europe. In the context of this volume, this chapter bridges the more detailed country surveys and the chapter on European-wide soil erosion assessments (Chapter 2.13). Therefore, the focus in scale is on national-scale data sets (i.e. data sets providing information for considerable parts of a country), which compared with the European-wide approaches are often based on larger scale data. In addition to documenting the wealth of data on soil erosion available on the national scale, we will try to answer the question of the extent to which these soil erosion data sets are comparable to each other. This question is strongly connected with the definition of the term soil erosion.

2.17.2 DATA ON SOIL EROSION

Soil erosion is a generic term covering all forms of human-enhanced soil removal by natural drivers (e.g. Bennett, 1939, p. 94; ISSS, 1996, p. 226; Shakesby, 2003), regardless of the specific processes detaching and transporting the soil.[1] Soil erosion by water (rainfall and snowmelt), soil erosion by wind and tillage and harvest erosion all contribute to soil erosion. In the area of water-driven soil erosion, it is additionally essential to distinguish between inter-rill and rill erosion, which can be studied on the plot scale, and gully erosion, which is observable only on the field or catchment scale (cf. Chapters 2.4 and 2.5).

A wide range of potential sources of information exist for soil erosion as a whole as well as many of the above-mentioned hierarchically organised sub-processes (Table 2.17.1). However, data derived from theses different sources are often referring to rather different temporal and spatial scales and are not easy to compare. Therefore, careful documentation of the concepts and methods involved, a stringent use of terminology and thorough consideration of the significance of the data are important when handling 'soil erosion data'.

The term soil erosion is seldom used in the description of the potential information summarised in Table 2.17.1. Instead, information is classified as data on either *soil loss* or *sediment yield*. Both of these well-established terms are used according to Wischmeier and Smith (1978), who themselves carefully avoided the term *soil erosion* in the discussion of the USLE (Universal Soil *Loss* Equation), the exception being the abstract of the publication. *Soil loss* [gross erosion according to van der Knijff *et al.* (2000, p. 23)] denotes the amount of material mobilised by erosive processes on a specific site (source area), neglecting possible deposition. With respect to the USLE, the term soil loss has an even more restricted sense, i.e. soil loss from inter-rill and rill erosion driven by rainfall only (Wischmeier and Smith, 1978, p. 5). In comparison with this, *sediment yield* denotes the amount of material delivered to a specific

[1]In contrast to this established meaning of the term soil erosion, the EEA and some publications by the JRC (e.g. Grimm et al., 2002a, p. 1/8) started to use the term soil erosion as a synonym for natural erosion [cf. definitions of soil erosion in ISSS (1996, p. 226) and EEA glossary of environmental terms, cited in Gobin *et al.* (2003, p. 96)]. Besides terminological confusion, this is unfortunate, as it causes misperceptions concerning the limitations of soil erosion models and undermines the possibility of distinguishing between soil erosion and natural erosion or geological erosion based on the classification of the source areas (e.g. van Lynden, 1995; Graf, 1996). If it is felt that the established term soil erosion for human-induced accelerated erosion is not specific enough, then the authors suggest following OECD (2003) and using 'agricultural soil erosion'. This should help to prevent expecting a soil erosion model such as the USLE to predict correctly soil loss in high mountain areas where gravity-induced slope processes (i.e. mass movements) dominate.

TABLE 2.17.1 Potential sources of information on soil erosion

Method	Spatial scale (km^2)	Temporal scale(years)	Type of soil erosion[a]	Potential information and limitations
Geomorphic and pedological analysis of landforms (e.g. lynchets)	10^{-1}–10^{0}	10^{1}–10^{2}	a, (b), c	Summarily assessment of past total soil loss for specific sites, often difficulties in dating start of development and no information on temporal variation of soil loss
Geomorphic and pedological analysis of slope profiles	10^{-1}–10^{0}	10^{2}–10^{3}	a, b, c	Summarily assessment of redistribution of soil on field sites, combination with dating methods yields intensities of soil redistribution, assessment of soil loss might be difficult
Investigations of fluvial environments (e.g. valley floors)	10^{1}–10^{6}	10^{2}–10^{3}	a	Summarily assessment of sedimentation along rivers, combination with dating methods yields former phases of extensive sediment transport (flooding), assessment of sediment yield rates extremely difficult
Investigations of lacustrine environments (e.g. closed hollows, lakes, reservoirs)	10^{1}–10^{2}	10^{1}–10^{3}	a	Summarily assessment of sediment yield amounts for specific catchments, combination with dating methods yields former sediment yield rates, annual resolution possible
Written accounts	10^{0}–10^{1}	10^{2}	a, b	Timing and causes of former soil erosion events, might include estimates of soil loss amounts
Mapping schemes	10^{1}–10^{5}	10^{0}–10^{3}	a, b, c	Information from mapping schemes strongly depends on the purpose of the mapping, ranges from *ad hoc* to long-term assessments of damage from soil erosion or specific types of soil erosion
Measurements on plots	10^{-4}	10^{-4}–10^{1}	a, b, c	Soil loss for specific sites with high temporal and spatial resolution, good control of system dynamics and processes involved is possible

(*Continued*)

TABLE 2.17.1 *(Continued)*

Method	Spatial scale (km^2)	Temporal scale(years)	Type of soil erosion[a]	Potential information and limitations
Measurements in catchments	10^1–10^6	10^{-4}–10^1	a	Sediment yield for specific points with high temporal and possibly spatial resolution, control of system dynamics and processes involved is possible, combination with plots measurements or mapping schemes might yield information about relation between soil loss and sediment yield (sediment delivery ratio)
Modeling exercises	10^{-4}–10^6	10^{-2}–10^1	a, b, c	Spatially distributed assessments of soil loss or sediment yield for specific types of erosion and specific temporal scales (medium-term means, event based). The authors are not aware of any model being able to assess soil erosion as a whole

[a]Types of soil erosion: a, soil erosion by water; b, soil erosion by wind; c, soil erosion due to machinery.

point at some distance from the source area (Walling, 1994). If deposition occurs between the source area and this specific point, then the sediment yield is smaller than the soil loss in the source area. However, if additional eroding sediment sources are present (e.g. bank erosion along water courses), sediment yield can also be higher than the soil loss in the source area (cf. Boardman *et al.*, 1990). Both measures, soil loss and sediment yield, can be considered as valid *indicators* of soil erosion, describing the state of the phenomenon (OECD, 1993), with *soil loss* being more important in the context of *on-site damage* and *sediment yield* being a better indicator for *off-site damage* from soil erosion. To summarise, neither of these two terms represents the full range of impacts of soil erosion on the environment and, even more important, they are not interchangeable.

Table 2.17.1 gives an overview of the potential sources of information on soil erosion and demonstrates the wealth of sources for assessing soil erosion in Europe on different spatial scales (from field to catchment) and over different periods (from single events to long-term mean values).

Geomorphic and pedological analysis of specific landforms (e.g. lynchets) and slope profiles has the potential to yield information on soil loss or soil redistribution over longer time-scales, varying from a few decades to several hundred years (Chapter 2.1). In order to derive total amounts or rates of soil loss, reconstruction of the former relief and dating of accumulated sediments are crucial. Owing to the intensive field and laboratory work involved in this kind of analysis, data for geomorphic and pedological landform analysis are often restricted to small areas (e.g. Bell and Boardman, 1992). An exception to this is the soil erosion assessment for Iceland (Arnalds *et al.*, 2001), where geomorphic field work and remote sensing were combined to give a unique mapping scheme for a nation-wide assessment of the overall on-site impacts of soil erosion since early settlement around 1100 years ago.

Investigations of fluvial and lacustrine sediments yield long-term information on sediment yield from catchments which can vary from a few to several thousands of square kilometres. Numerous studies have shown that the onset of soil erosion in a region and episodes of over-bank sedimentation (i.e. episodes of intense soil erosion) can successfully be derived from investigations of alluvial stratigraphy (e.g. Needham and Macklin, 1992). However, owing to the frequent reworking of river deposits and difficulties in determining even present day sediment delivery ratios of fluvial systems (Asselman, 1999), sediment loss estimates for a catchment area based on fluvial sedimentation records are hard to find. Owing to the well-defined sediment trap efficiency of lakes and reservoirs (Heinemann, 1984), investigations of lacustrine sediments are much more promising and have been successfully used to estimate sediment yield and soil loss (after taking into account sediment delivery ratios) from catchments for periods ranging from a few decades (van Rompaey *et al.*, 2003) to centuries (Dearing, 1994). In many European countries, however, lakes and reservoirs are mainly located in upland regions and data derived from these locations often tell more about natural erosion than human-induced soil erosion.

Current measurements of soil erosion based on mapping schemes, plot measurements or sediment yield measurements in catchments have the advantage of potentially recording not only amounts of soil loss or sediment yield but also the complex environmental conditions controlling soil loss and sediment transport. Until recently, mapping schemes were mainly restricted to short-term efforts covering often small regions (cf. GCTE Focus 3 Office, 1997; Favis-Mortlock, 2003; cf. Table 2.17.2). Here, the multi-purpose land cover and land use statistical survey (LUCAS) marks a new approach in Europe (Bettio *et al.*, 2002). Based on an EU-wide area frame sampling strategy, spot observations on soil erosion (i.e. rill and gully erosion) and accumulation are collected on arable land and used as indicators for on- and off-site damage, respectively. The first results were disturbing, as they showed, for example, for Germany strong erosion in areas with low soil erosion risk and only little erosion in high-risk areas (Eiden and Bettio, 2002). However, owing to the strong year-to-year spatial variation of soil erosion events, these first single-date observations should not be overestimated. Despite some practical and conceptual shortcomings, the value of the LUCAS approach lies clearly in the intended frequent repetition of the mapping exercise and the large number of observation points.

Soil loss measurements on the plot scale, ranging from small laboratory plots to plots covering a hill slope, have a long tradition in soil erosion research. Data on soil loss measurements from plots have been published for most European countries, especially in the 1970s and 1980s. Owing to specific research questions or for practical reasons, plot design often varies considerably and comparison of the data is not straightforward. Another problem is the access to the data, which often have been published in 'grey' literature. In the mid-1990s, the GCTE Focus 3 summarized metadata for plot studies from around the globe (GCTE Focus 3 Office, 1997). The updated version (Favis-Mortlock, 2003) lists current metadata from about 10 sites across Europe. In addition, the European Society for Soil Conservation (ESSC) set up a task force to create a data bank on erosion plot studies in 1998. Unfortunately, we were not able to locate any publication following this initiative.

From the mid-1980s on, when the focus of European soil erosion research shifted towards off-site impacts of soil erosion and GIS technology became available to handle complex spatial data, soil erosion studies at the catchment scale became increasingly common. When catchment studies are designed in a nested catchment approach and combined with mapping schemes, the results derived from these studies are extremely valuable since they provide information on soil loss and sediment yield dynamics under current land-use conditions (e.g. Takken *et al.*, 1999). Most catchment studies in the context of soil erosion studies still cover, however, rather small areas of a few square kilometres and only short periods of a few years (cf. GCTE Focus 3 Office, 1997; Favis-Mortlock, 2003). This gap can partly be closed with sediment yield data from national water authorities, which in some countries take regular measurements in larger rivers (e.g. Asselman, 1999).

Finally, modelling approaches can be used to create data sets on soil erosion for specific areas and time frames. Depending on the model applied (cf. Chapter 2.16), data differ in (i) the particular soil erosion process taken into account (soil erosion by water from inter-rill and rill erosion, soil erosion due to snowmelt, soil erosion by wind, etc.), (ii) the parameter that reflects soil erosion (soil loss or sediment yield) and (iii) the time frame (single event to means over several years). In addition to these obvious differences in soil erosion data sets derived from modelling approaches, the spatial resolution of the input data influences model output fairly strongly. For example, for USLE-based data sets the predicted soil loss is very sensitive to variations in slope angle, i.e. a parameter strongly dependent on the grid size of the elevation model used (cf. van Dijk and Kwaad, 1999, pp. 15*ff.*).

2.17.3 SOIL EROSION DATA SETS ON THE NATIONAL SCALE

Several publications have stressed the patchiness of reliable measurements of soil erosion (e.g. EEA and UNEP, 2000, p. 21) or inadequate input data for soil erosion assessments in Europe (e.g. van der Knijff *et al.*, 2000, p. 9). The COST Action 623 'Soil Erosion and Global Change' (1998–2003) started from a different point of view. It acknowledged the wealth of data on soil erosion in Europe, being aware that much useful data had been collected for specific purposes and therefore differ in spatial and temporal resolution and meaningfulness. In addition, it was recognized that the data are rather dispersed, with language barriers often impeding the European-wide perception of studies aimed at a national or regional scale or audiences, respectively.

Table 2.17.2 summarises metadata on national-scale soil erosion assessments from European countries (the biased selection of the countries reflects the participation in Working Group 3 of the COST Action 623). For all listed countries, some kind of nation-wide data sets on soil erosion are available. The common scope of these data sets is to assess comprehensively the extent and magnitude of soil erosion and to identify areas with strong soil erosion in order to identify regions where more precise evaluations are necessary and to facilitate sustainable land utilization (see references in Table 2.17.2). In Iceland, the assessment of soil erosion is based on the overall long-term impact of soil erosion, i.e. the *status quo* of soil degradation due to soil erosion. In all

TABLE 2.17.2 Metadata on national-scale data sets on soil erosion in Europe

Country	Type of erosion[a]	Method	Resolution	Scale	Reference
Austria	a	Expert knowledge	Not applicable	Not applicable	BFL (1997)
Belgium	a	USLE[b]	20 m	?	Flanders only van Rompaey (2000)
Czech	a	USLE[b]	50 m grid	1:200 000[c]	Dostal *et al.* (2002)
Republic	b	WEQ	Not applicable	1:800 000	Pasak and Janecek (1971)
Denmark	a	USLE[b]	50 m grid	1:50 000	Olsen and Kristensen (1998)
	b	Mapping	Not applicable	?	Kuhlman (1986)
Finland	a	USLE[b]	200 m	1:1 000 000[c]	Laine and Rekolainen (1996)
France	a	INRA	250 m grid	1:1 000 000	Le Bissonnais *et al.* (1998)
Germany	a	USLE[b]	50 m grid	1:2 750 000	Hennings (2003)
Hungary	a	USLE[b]	?	1:100 000	Centeri and Pataki (2000)
Iceland	a, b	Mapping	Not applicable	1:100 000	Arnalds *et al.* (2001)
Italy	a	USLE[b]	250 m grid	1:1 000 000	Grimm *et al.* (2002b)
Norway	a	Mapping	Not applicable	?	Leek and Rekolainen (1996)
Poland	a, b	Expert	Not applicable	1:300 000	Józefaciuk and Józefaciuk
		knowledge	Not applicable	1:500 000	(1995)
Portugal	a	RUSLE	?	?	Chapter 1.28
Slovakia	a	USLE[b]	50 m	1:500 000	Suri *et al.* (2001)
Spain	a	ICONA	?	1:400 000	ICONA (1987*ff.*)
England	a	Mapping	Not applicable	1:250 000	Evans (1985)
and Wales	a	Mapping	Not applicable	?	McHugh *et al.* (2002)

[a]Types of soil erosion: a, soil erosion by water; b, soil erosion by wind.
[b]USLE-based soil erosion assessments are not fully comparable, as USLE factor estimations are often modified to meet the limitations of available input data; see references for details.
[c]Assessment of scale based on smallest scale of input data.

the other countries, present-day soil erosion rates are used as an indicator. However, even in these countries the type of soil erosion considered, the methodology to derive data on soil erosion and the scale or resolution of the data sets differ.

Comparing the types of soil erosion considered, there is a clear dominance of data on soil erosion by water, which is available for every country listed. An exception is Iceland, where no distinction was made between soil loss by water and soil loss by wind. Additional nation-wide assessments of soil erosion by wind are only available for the Czech Republic, Denmark and Poland. However, the lack of data on wind-driven soil erosion is partly an artifact of the scale considered, i.e. the national scale, thus omitting existing assessments which cover selected regions where wind erosion is considered to occur (e.g. northern Germany, eastern England; cf. Chapter 2.7).

Regarding methodology, most nation-wide soil loss data sets (in nine out of 16 countries) rely on USLE-type modelling approaches to assess the extent and spatial variation of soil erosion by water (inter-rill and rill erosion only). This is due to the limited demand of the USLE for input data and the lack of input data on a national scale to run more sophisticated soil erosion models (cf. de Roo, 1999). In most countries that apply USLE-type models, plot studies have been conducted to adapt the factors of the empirical soil loss equation to the specific environmental conditions such as soil properties and climatic conditions (for details, see references in Table 2.17.2). These efforts are as yet not comparable to the work performed in the USA. Apart from the

known shortcomings of the USLE, rigorous long-term plot study validation over a range of environmental conditions remains an issue for justifying its application in Europe and its implementation into policy guidelines. There are still doubts about the accuracy of the soil loss amount predicted by USLE (Burwell and Kramer, 1983; Govers, 1991; Risse *et al.*, 1993; Evans, 1995; Baade *et al.*, 1996). In addition to the USLE, specific multi-factorial modelling approaches have been utilised in France and Spain. These approaches basically take into account the same driving forces for soil loss as the USLE-based models. However, the calculations are based on classified input data instead of metric-scaled input data. Mapping schemes provide the database for Iceland, Norway and the UK [England and Wales only, arable land by Evans (1985) and uplands by McHugh *et al.* (2002)]. In Austria and Poland, expert knowledge has been used to identify regions affected by soil erosion by water. Data sets on soil erosion by wind are based in the Czech Republic on a modelling exercise, in Denmark on a mapping scheme and in Poland on expert knowledge.

As far as modeling approaches and digital data sets are concerned, the median resolution for pixel-based calculations of soil loss is 50 m, ranging between 20 m for Flanders and 250 m for France and Italy. In many cases, the pixel size is based on the input data set with the highest resolution, i.e. often the digital elevation model. This might be justified by the high sensitivity of soil loss to slope angle, but it blurs the fact that in most countries soil data sets covering the whole country are based on soil maps in the scale of 1:1 000 000, a scale 'regarded as of little value for in-country management of soil resources' (Bullock, 1999, p. 17). Only for Belgium, The Netherlands and Denmark are complete soil surveys at the 1:50 000 scale or larger available. The lack of high-resolution national soil data sets has the advantage that many national soil loss assessments are based on comparable soil data. The same is true for the land-use data utilised in these modelling approaches, which generally are based on the 1:100 000 scale CORINE land cover data set. However, the CORINE data set only distinguishes between arable land and several other agricultural and non-agricultural land-use types (Heymann *et al.*, 1994). In order to assess actual soil loss, information on crops grown and ideally on cropping methods is needed. Whereas information on grown crops is readily available from regionalised agricultural statistics, although not always utilised, information on cropping methods are generally lacking. This has to be considered a major data gap for any assessment of actual soil loss, because improved land management practices have the potential to, e.g., reduce USLE soil loss for a given crop by a factor of five (cf. Schwertmann *et al.*, 1987, p. 49). Another feature not included in present-day nation-wide soil loss modelling approaches is soil protection measures, such as terraces or traditionally reduced field sizes (slope lengths) in hilly terrain. In many countries that utilise USLE-based modelling approaches, considerable efforts have been undertaken to assess the spatial variability of rainfall erosivity. In fact, within the group of model-based soil erosion assessments in Table 2.17.2, the data set for the Czech Republic is the only one based on a constant erosivity factor for the whole country (cf. Dostal *et al.*, 2002).

Summarising the brief evaluation of input data sets used in national modelling approaches to assess within-country spatial variation of soil loss by water, one can conclude that most data sets are based on similar methodology and input data sets and often have similar drawbacks. From this point of view, a large number of national data sets on soil loss are comparable to each other and might well provide a higher resolution database for comparisons with methodologically similar European-wide soil loss assessments (van der Knijff *et al.*, 2000) in order to investigate scale effects resulting from the utilisation of input data sets with lower resolution. However, this requires that the results from these modelling approaches are represented in a similar manner. The probably most heterogeneous property of national data sets on soil erosion is the representation of soil loss amounts. Many maps representing these data sets just show intensity classes of soil loss (often five classes from very low to very high), without providing any information on the underlying classification of the metric data derived as model output. Even if it is argued that a qualitative ranking is sufficient to identify regions where more detailed studies are necessary, this representation of the model results hinders evaluation of the data sets and comparison with data on soil loss from other sources. Even where quantitative data are provided, or might be traced back, very different classification schemes are applied (cf. soil loss maps for Germany,

Hungary, Italy, the Czech Republic). One step in harmonising national data sets could be to agree on a common classification of the data, which is possible as most data sets are stored in digital form. Regarding the representation of model results, most USLE-based data sets include soil loss estimates for the entire country or for all land-use classes. Where representation of data is based on an aggregation of results at administrative unit levels, this can be explained by the methodology. However, most data sets are produced at fairly high resolution. Therefore, it should be possible to mask all the areas where model application is not valid (e.g. forested areas, high mountain areas for the USLE). A good example is the recently published soil loss risk map of Germany (Hennings, 2003).

2.17.4 CONCLUSIONS

In contrast to the often stated lack of information on soil erosion in Europe, this chapter – as well as this volume – demonstrates the wealth of national-scale information on soil erosion. For every country considered in this review, some kind of national-scale data sets on soil erosion are available. The common scope of these data sets is to assess comprehensively the extent and magnitude of soil erosion and to identify areas where more precise evaluations may be necessary to facilitate sustainable land utilisation. We have shown that the data sets differ in methodology and the soil erosion processes considered and that they are not fully comparable. The majority of national-scale data sets are based on USLE-type modelling approaches, providing information on either actual or potential current soil erosion by inter-rill and rill erosion from rainfall, thus omitting any other processes contributing to soil erosion. In other countries, expert knowledge, mapping schemes and other kinds of (factorial) modelling approaches have been used to assess primarily soil erosion from rainfall. Nation wide data sets on soil erosion by wind are available for the Czech Republic, Denmark and Poland.

Compared with the situation at the end of the 1980s, when the first European-wide assessment of soil erosion resulted in a map identifying hot-spots of soil erosion (de Ploey, 1989), the present situation marks a major step forward. However, this progress and the vast amount of information available should not distract from the still existing shortcomings in the available data sets on soil erosion as pointed out in this chapter.

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2.18

Impacts of Environmental Changes on Soil Erosion Across Europe

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2.18.1 INTRODUCTION

Soil erosion rates are influenced by both physical and socio-economic factors, operating at a wide range of temporal and spatial scales (Figure 2.18.1). In this chapter, we explore impacts of environmental change on soil erosion by water, first through a discussion of the processes of erosion and the mechanisms by which they are modified, directly and indirectly and second through the incorporation of these mechanisms in a coarse-scale model which allows an analysis of the combined impact of some relevant factors, taking account of their mutual interactions.

The principal drivers of environmental change are land use and climate. Although climate change may currently be proceeding at a higher rate than ever before, the direct impact of land-use change in Europe may be even stronger, under economic and demographic pressures which are changing the face of southern and eastern Europe. In addition, there is a knock-on effect of global climate change on socio-economic conditions which may drive further land-use change, although the directions of change are far from clear.

In this chapter, the impacts of environmental change, and in particular changes in climate and land use, are explored through specific reference to the driving mechanisms, and so through an understanding of exactly how an environmental change is responsible for a change in soil erosion rates. As will be seen in the discussion below, these mechanisms act through changes in the hydrology of runoff generation and the hydraulics of sediment transport. Climate change directly and immediately impacts rainfall intensity (Figure 2.18.1), and more slowly modifies soil characteristics, acting both via vegetation growth and via the effects of cumulative erosion and associated armour development. Land-use change also affects the hydrology, primarily by influencing the periods of the year when the soil surface is exposed to crusting and consequent enhanced runoff, but also by modifying surface roughness and soil organic matter.

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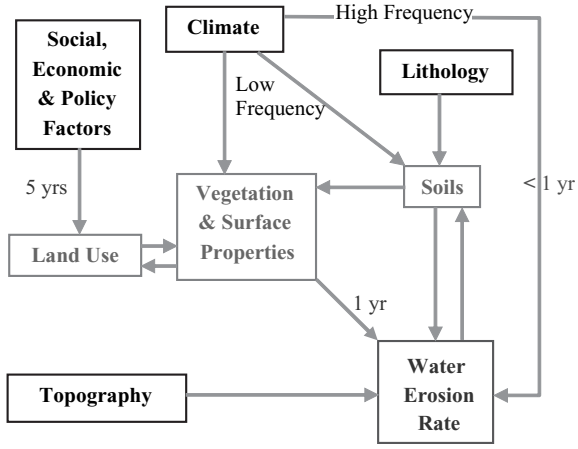

Figure 2.18.1 Factors influencing water erosion rates. Figures indicate order of magnitude for response times

The most immediate impact on soil erosion is the effect of individual storms and any map of actual erosion is strongly linked to the incidence of particular, and often highly localised, large rainfall events. Both the relationship between rainfall and runoff and the relationship between runoff and erosion are highly non-linear, so that the small areas with the largest and most intense rain produce far the greatest soil erosion. Where this storm centre is a part of a larger area event, the eroded soil may travel far downstream, but it is more usual for most of the material to be redeposited locally, at the bottom of fields and in the muddy floods which can cause extensive damage to farms and neighbouring settlements. Other factors can, however, greatly modify both the runoff and erosion resulting from a given storm.

Other important and immediate influences on storm erosion are the surface properties and vegetation. Protection by a crown cover breaks the impact of falling raindrops and largely prevents the formation of a surface crust on susceptible soils, particularly those with a high silt content. Crusts are formed by breaking soil aggregates down to their constituent grains, which then cement remaining aggregates to form a largely impermeable seal. This seal greatly reduces the rate of infiltration and so promotes runoff and erosion. For a given runoff rate, vegetation stems also reduce erosion by providing frictional roughness which slows the flowing water and absorbs its momentum. In the longer term, the vegetation grows in response to the seasonal course of the weather but is, in many places, also subject to cultivation, grazing, fire and other predominantly human influences. Where natural vegetation develops, often over many decades, it produces and maintains soil organic matter. This improves the water-holding capacity of the soil and further increases infiltration, so that areas of mature forest, for example, rarely produce runoff and erosion.

Land-use choice is the end product of an interplay between economic conditions, the regulations in force, social circumstances and the physical constraints on crop growth. Changes in climate are one factor in this complicated and dynamic system, influencing both local physical conditions and global market prices. Land use directly controls the annual pattern of vegetation cover and inappropriate choices can greatly increase erosion losses, particularly where the land is left unprotected by plant cover at times of year with severe rainstorms. Many other aspects of cultivation practice may also influence erosion rates, including the frequency and direction of tillage, management of headlands and wheel tracks and protection of areas at greatest risk.

Soils also influence erosion rates through the grain size of the soil particles and structures developed over time due to biological exchanges, water flow, loss of nutrients by leaching and renewal from the weathering of parent materials. These factors act physically by promoting crusting in silty soils and by promoting protective armouring where stony soils erode, chemically by providing the natural level of nutrients for plant growth

and biologically through organic nutrients and the water retention of organic matter. Where erosion is not in balance with the rates of renewal by weathering and by net sequestration of organic carbon, soil properties gradually change, and in turn modify the runoff and erosion rates. In the long term, erosion can eventually modify the entire landscape, changing the mosaic of soils in an area, stripping areas down to bedrock and potentially fragmenting the land by cutting gullies.

Environmental changes can modify many of these processes, following physical chemical and socio-economic pathways, many of which are incompletely understood. Here we focus primarily on the physical processes, but these must be set within the context of the socio-economic drivers which influence land use, often without giving high priority to the short- and long-term erosional consequences.

2.18.2 DRIVING MECHANISMS FOR SOIL EROSION

Sediment transport erosion is directly driven by overland flow runoff and slope gradient. A simple approximation (Carson and Kirkby, 1972) is

$$q_s = kq_w^2 s \tag{2.18.1}$$

where q_s = sediment discharge per unit width, q_w = water discharge per unit width, k = soil erodibility and s = slope gradient.

This expression encapsulates much of what we understand about soil erosion and the impacts of environmental change on it. What is immediately apparent from the nonlinearity of this expression is that the greatest impacts on erosion rates are through changes in the water discharge, although lesser impacts can occur through changes in gradient or soil erodibility. The next stage of understanding the impacts on erosion therefore lies in the mechanisms which control overland flow discharge. Overland flow can occur either because rainfall intensity exceeds the infiltration capacity [Hortonian overland flow (HOF)] or because the soil is saturated [saturation overland flow (SOF)]. All overland flow generally erodes some soil, but the effects of erosion are much greater where the flows of water and sediment connect together and build up over a larger area.

2.18.2.1 Hortonian Overland Flow

For HOF to occur, runoff must first be generated locally and second connect with other areas of runoff to build up a significant discharge. Storm rainfall is partitioned (Figure 2.18.2) by the vegetation canopy and at the ground surface. A small amount of rain is intercepted by vegetation and physically held on leaves and stems, where it will remain until it drains or is evaporated. Where the crown cover is not too sparse, a much larger fraction of the rainfall strikes the canopy, which absorbs its kinetic energy, so that rain strikes the ground with only the energy of falling from the plant leaves at velocities of about $1\ \mathrm{m\ s^{-1}}$, instead of the much higher energy of free fall from the clouds at up to $10\ \mathrm{m\ s^{-1}}$.

The process of infiltration is far from simple, owing to the heterogeneity of the soil. Surfaces may be crusted, wholly or in patches, and generally consist of fine aggregates separated by a structured three-dimensional network of macro-pores which carry much of the infiltrating flow. A complete physical representation is elusive, and simpler conceptual models are usually applied. The simplest approach is to use a 'leaky bucket' model, in which all rainfall infiltrates until the bucket is full, and thereafter a fixed proportion of subsequent rainfall infiltrates (de Ploey *et al*., 1991), while the remainder runs off as overland flow. The total accumulated storm runoff J may therefore be written as

$$J = p(r - h) \quad \text{if} r \geq h, \text{ zero otherwise} \tag{2.18.2}$$

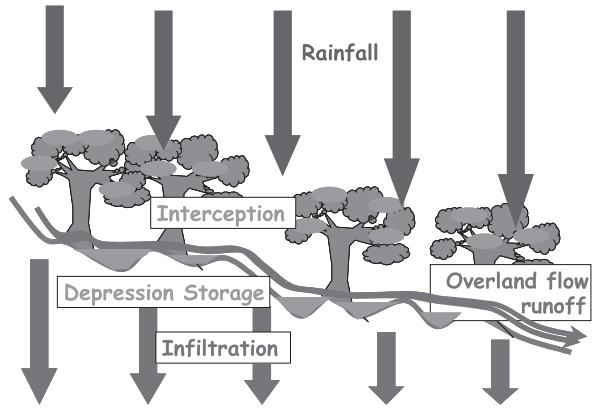

Figure 2.18.2 Partition of storm rainfall between canopy, overland flow and infiltration

where r is the accumulated storm rainfall, h is the runoff threshold (i.e. the bucket size) and p is the fraction of subsequent runoff.

This form of expression has the advantage of conceptual simplicity, but shows no direct response to variations in storm intensity, which are often associated with observed differences in runoff amounts. However, intensity data are often not available, so that this apparent weakness may be a practical advantage.

A second approach is the SCS curve number approach (USDA, 1972; Haan *et al*., 1994). Using the same terminology as above,

$$J = (r - h)^2/(r + 4h) \tag{2.18.3}$$

where the SCS curve number $C = 1000/(10 + 0.2h)$ for h in mm and $h = 0.2SS$ for comparison with the storage SS used in the original SCS literature. This family of curves is empirically based on runoff experiments, is even simpler than the bucket model and shares with it an absence of intensity dependence.

A third approach is to use an infiltration equation, and associate the equation with an effective steady rainfall intensity, i_*. For the Green–Ampt equation, which has previously been compared with the curve number approach (Nearing *et al*., 1996; Smemoe, 1999), infiltration is estimated as

$$f = A + B/S \tag{2.18.4}$$

where f is the infiltration capacity, for constants A and B, and the storage S is interpreted as the amount of water which contributes to the tension gradient. This maintains consistency with conditions for horizontal infiltration, and is consistent with the Philip infiltration equation for maximum infiltration capacity:

$$f = A + \sqrt{B/2}\, t^{\frac{1}{2}} \tag{2.18.5}$$

Ponding occurs at time

$$t_P = B/(i_* - A)^2 \tag{2.18.6}$$

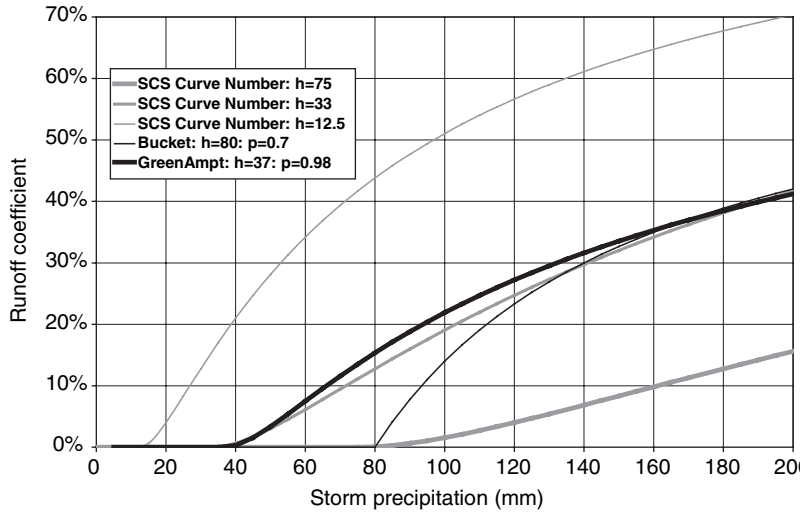

Figure 2.18.3 Comparison of bucket, SCS curve number and modified Green–Ampt as storm runoff models

At this time the storage $S = B/(i_* - A)$ and cumulative rainfall up to ponding is $i_* t_* = Bi_*/(i_* - A)^2$. After ponding has developed, the subsequent total runoff is

$$J = ph\left[\frac{r}{h} - \sqrt{\left(2\frac{r}{h} - 1\right)}\right] \tag{2.18.7}$$

where $p = 1 - A/i_*$ and $h = Bi_*/(i_* - A)^2$.

All three methods have a clear lower threshold for runoff, and asymptotically tend to 100 % runoff in large storms (for $p = 1$). Although there are some differences in detailed form, there is good agreement within the range of experimental error, as is illustrated in Figure 2.18.3.

The advantage of the Green–Ampt formulation is that it is empirically adequate, taking a very similar form to the curve number method, is physically meaningful because it can be interpreted directly in terms of the infiltration parameters and has the advantage that it can make use of effective rainfall intensity where this is available. This is important for improving forecasts of runoff during the course of a storm, and has the advantage of not depending on the choice of exactly when each storm is said to begin. Application of the Green–Ampt or other infiltration equations clearly demonstrates that short-term intensity variations within a storm are very important for local runoff generation. However, although it is likely that climate change is likely to change intensities, there is little clear evidence on probable trends at present.

Hortonian overland flow may be generated anywhere within a catchment, but occurs preferentially on areas with soils and land use which give low runoff thresholds. It has been seen that runoff is increased by sparse vegetation, particularly where soils are prone to crusting, and that land use is a major factor in determining vegetation cover and its seasonal pattern, within the constraints set by water availability for plant growth. HOF is widespread in semi-arid areas, either throughout the year or during the dry season, but also occurs under arable crops where the cultivation calendar leaves the soil bare at times of year when there is a risk of storms. Topography plays only a minor role in controlling the spatial pattern of Hortonian overland flow, except on steep slopes where thin soils limit infiltration and give low runoff thresholds.

2.18.2.2 Saturation Overland Flow

Under humid conditions, with rainfall exceeding potential evapotranspiration for all or part of the year, most runoff generally occurs by lateral flow within the upper layers of the soil. Lateral hydraulic conductivity and porosity normally decrease substantially with depth, both under natural conditions with the incorporation of organic matter and through tillage. Rain percolates downwards until it reaches less permeable and/or saturated layers, and then flows laterally. These lateral flows accumulate with drainage area, so that increasing volumes of water flow through points further downslope, leading to wetter conditions and perhaps eventually to saturation. Subsequent rain, even at low intensity, cannot enter the saturated soil and flows off as saturation overland flow.

For a steady-state condition, then the discharge for a continuous net rainfall intensity of i is given by

$$q = aj = ai = T(D)s \tag{2.18.8}$$

where q is the discharge per unit contour width, a is the area drained per unit contour width, j is the runoff (per unit area), D is the local soil profile deficit below saturation, $T(D)$ is the transmissivity of the soil, a decreasing function of the soil deficit D, and s is the local water table gradient, which is close to the topographic gradient except on the gentlest slopes.

It can be seen that as the 'topographic index', a/s, increases downslope then the transmissivity must also increase and the deficit must fall. A critical value can be defined for a/s, corresponding to the seasonal meteorological conditions and soil texture, for which the deficit falls to zero, corresponding to saturation of the soil profile. If a/s is mapped over an area, the region with more than the critical value for saturation defines the area generating SOF. The topographic index increases steadily downslope, but increases much more rapidly in hollows, with flow convergence in plan (increasing a) and on profile concavities (increasing s). The areas of SOF production are therefore very strongly structured by catchment topography.

Since subsurface flows are driven by a positive net rainfall, then SOF only occurs where rainfall exceeds potential evapotranspiration for long enough for the catchment to approach steady-state flow conditions. This equilibrium time increases as the mean subsurface runoff decreases, so that the conditions for SOF become increasingly rare along a climatic gradient from humid to arid conditions. In humid areas, SOF dominates year round. Transitions towards a more Mediterranean regime show seasonal shifts between regimes, SOF in winter and HOF in summer, and in more arid areas only exceptional wet periods can trigger SOF.

2.18.2.3 Connectivity

Not all of the overland flow generated necessarily reaches a flowing stream channel, but may reinfiltrate en route. If it does so, it deposits all of its sediment load. Alternatively, the flow may be redistributed across an area, with or without some reinfiltration, and lose some of its transporting capacity, again leading to some deposition of transported sediment. Connectivity describes the probability distribution that material (water and sediment) that will flow between an upstream/upslope point and a downslope/downstream point and the average connectivity for sediment in relation to the catchment outlet is the sediment delivery ratio. In many catchments, sediment connectivity is low, with much eroded material redeposited at the base of fields and on flood plains. Much of the sediment, particularly coarse sediment, in streams is immediately derived from bed and bank erosion, so that the processes of hillslope erosion and channel sediment transport may be strongly decoupled in time and space, only connecting together over the time span in which the flood plain is reworked by its river. Where flow connectivity is maintained, local deposition of sediment in plough furrows, on terraces and buffer strips or behind check dams may be compensated by fresh erosion downstream as soon as the flow recovers its original gradient and transporting capacity.

For HOF, flow connectivity is constrained by patchiness of the land cover and by the duration of intense bursts of rainfall within a storm. Patchiness occurs at all scales. At the fine scale, shrubs may protect areas of high infiltration with runoff generated on the bare patches between. At coarser scales, patches may consist of fields with different land use. Patches of high infiltration are usually provided by denser vegetation, and these can disconnect the flow naturally or through deliberate creation of buffer strips. Loss of connection is also provided by the structure of rain storms, which generally include short bursts of intense rain, punctuated by much longer periods of lower intensity. Overland flow is generated everywhere during an intense burst, but begins to flow downslope, generally at velocities of 1–10 cm s^{-1}. When the rainfall intensity falls, this flowing water is able to reinfiltrate, so that the stream only receives overland flow generated close to it. For example, flow generated during a 10-min burst will only travel 30 m (at 5 cm s^{-1}) before the end of the burst. Connectivity therefore declines strongly with distance from the nearest channel. Where the runoff threshold is lower, the duration of bursts intense enough to exceed the threshold is longer, so that connectivity is greater. Pursuing this argument, both runoff generation and connectivity generally increase with storm magnitude and with decreasing runoff thresholds.

For SOF the pattern of connectivity is considerably different. Overland flow is generated locally wherever the topographic index (TI) is greater than the currently critical value, which changes through a storm. If the downslope path for this water crosses areas with a lower TI, then the water can reinfiltrate and flow subsurface. The critical condition for connected overland flow is then that the minimum TI along the flow path exceeds the critical value, i.e. that the entire flow path is saturated. In a landscape for which the topography is more or less in equilibrium with prevailing geomorphological processes, the topographic index generally increases steadily downslope, so that any area generating SOF will always be connected to the outflow, but where catchments are not in equilibrium, for example with residual summit plateaus of locally incised gullies, then there may be areas where the TI decreases downslope, owing to anomalous convexities, and disconnected areas of saturation may occur. Hence the pattern of connectivity is determined by topographic anomalies, but not structured in a simple way.

In both humid and arid environments, roadways and other linear features are important, especially in humid areas where they may be main bare areas and therefore sources of overland flow. In forested areas, their impact on runoff and erosion can be dominant where they are well connected with outlets, but less important where overflow points route water back into well-vegetated areas.

Buffer strips have the opposite effect, catching water and/or sediment, but it is important to ensure that the water infiltrates or is diverted to low gradient pathways, to prevent fresh erosion downstream where the flow recovers its transporting capacity on steeper gradients and/or with more erodible materials (clear water effect).

It is plain from this discussion that environmental change can impact strongly on runoff, and therefore on erosion. Climate change can operate at many levels and, although it is widely assumed that anthropogenic impacts are already modifying temperature and total precipitation, the effects on storm frequency and within-storm intensity profiles are still uncertain. It is clear that changes in the intensity and duration of intense showers within a storm will modify the generation and delivery of HOF, and will have its greatest impact in semi-arid areas, but will also have a direct effect on SOF. Changes in average temperature, with or without changes in average monthly rainfalls, will have a stronger impact on subsurface flow, and so will be more important in humid areas, but may also change the boundaries between areas of HOF or SOF dominance.

Land use change has a strong impact under all conditions, and is likely to have a stronger influence than climate change on soil erosion in Europe. There is, however, some scope to mitigate its effects and the impact of climate change, through reduced tillage, other physical conservation measures and a careful choice of areas for creating set-aside and buffer strips in strongly affected regions. Reduction of overland runoff may also have beneficial effects in reducing loss of nutrients and pollutants, in combination with other methods.

2.18.2.4 Erosion Rates

One aspect of intensity which can have an important impact on runoff and erosion is the distribution of storm (or daily) rainfalls, and there are good prospects that GCM forecasts will soon be able to address this issue. To estimate the relative impact on total erosion of various intensity factors, the contribution from storms of different sizes must be weighted by their relative frequencies. Daily rainfalls for any month can be fitted well to a gamma distribution (Kirkby *et al.*, 2000), specified by the mean rain per rain day and its coefficient of variation. The probability density for a rain day of r is given by

$$p(r) = \frac{(\alpha r/\overline{r})^{\alpha-1}\exp(-\alpha r/\overline{r})}{(\alpha/\overline{r})\Gamma(\alpha)} \tag{2.18.9}$$

where $\overline{r}$ is the mean rain per rain-day and α is a parameter of the distribution, related to the coefficient of variation (CV) of rain on rain-days by $\alpha = 1/(CV)^2$.

Applying the simple bucket model of Equation (2.18.2) to daily rainfalls, and summing across this frequency distribution, the total relative erosion has been estimated in Figure 2.18.4 for a runoff threshold [h in Equation (2.18.2)] of 30 mm, and can be seen to increase strongly with the mean rain per rain-day (keeping total rainfall constant), and with increased variability of the daily rainfalls, associated with greater storminess. These increases are very much larger than those associated with increases in total rainfall by increasing the number of rain-days without changing intensity, also indicated in Figure 2.18.4.

The impact on runoff and erosion of the runoff threshold, which integrates the effects of land cover and soil properties, is illustrated in Figure 2.18.5 for a total annual rainfall of 500 mm. The runoff thresholds shown correspond roughly with bare areas ($h = 10$ mm), grass ($h = 40$ mm) and forest ($h = 100$ mm). It can be seen that impacts through land use can also be very strong, and that land use provides a critical mechanism in mitigating the impacts of harmful climate change, forming the basis of much conservation practice. In fact, the differences due to land cover are likely to be even stronger than shown in Figure 2.18.5, since erodibility, held constant in the figure, generally decreases as vegetation cover becomes denser.

The runoff threshold can also be directly modified by tillage practices which increase depression storage (Figure 2.18.2), either by ploughing along the contour or by constructing terraces, and such techniques can be

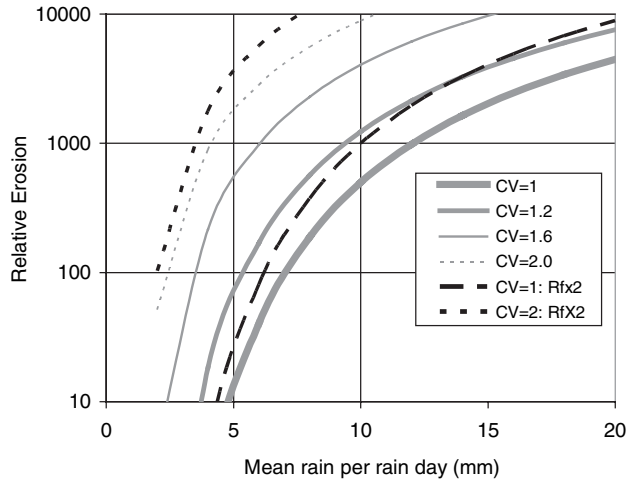

Figure 2.18.4 Relative erosion rates in relation to mean rain per rain day, its coefficient of variation (CV) and total rainfall. High CV is associated with a greater frequency of both lowest and highest storm rainfalls

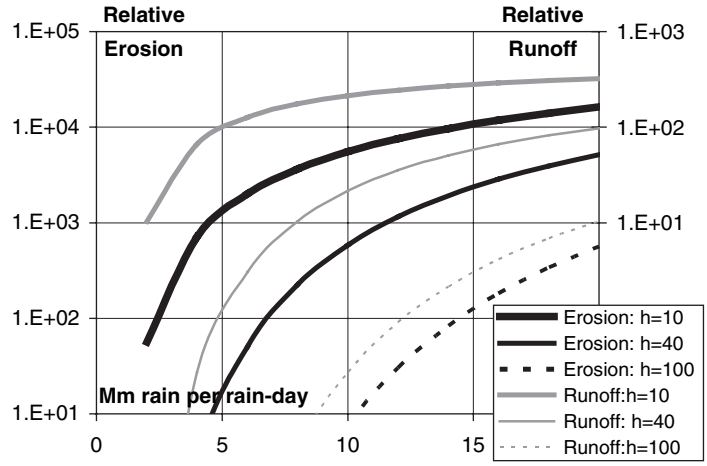

Figure 2.18.5 Effect of runoff threshold (h) on relative rates of total runoff and erosion as mean rain per rain-day is increased

used to reduce runoff and therefore erosion. However, if the depressions can be overtopped in a large rainstorm, much of the stored water is released as the lip of the depression is eroded, and erosion in an extreme event may actually be greater than without these conservation practices. For terraces, potential overflow routes can be identified and protected with stones or vegetation. Ponded water on terraces can also give rise to undermining of the terrace risers by piping where the soils are dispersible.

2.18.3 THE PESERA/RDI MODEL

Although the process discussions above give clear indications about the direction and extent of impacts on soil erosion due to changes in climate and land use, it is also instructive to place these trends in a geographical context, and relate them to the spatial pattern of current climate change scenarios for the next 50–100 years. Here we apply a slightly simplified process model, with assumptions that are compatible with the discussion above, that has been parameterised at 1-km. resolution for the whole of Europe. This is the PESERA/RDI model, which has been developed within a series of EU projects (Kirkby *et al*., 2000; Govers *et al*., 2002). Figure 2.18.6 shows the process components within the model. Runoff production is represented by a bucket model [Equation (2.18.2)], without any explicit correction for connectivity effects. Storage capacities are estimated from textural characteristics derived from the European Soils Database, modified by 'crustability', also based primarily on texture. Total effective bucket storage (runoff threshold) is estimated as a weighted average of the contributions from vegetated and bare soil proportions, with crusting influencing bare areas and subsurface saturation setting an upper limit on the runoff threshold for both bare and vegetated patches. Additional factors are included in the model for cold conditions, to allow for snow cover, snowmelt and frozen ground, but these are not discussed further here.

The distribution of daily rainfalls is represented by a gamma distribution for each month separately, using parameters derived from daily rainfall data, interpolated to a 50-km grid in the MARS database. Topographic factors are subsumed within the local relief, estimated as the standard deviation of elevation with a 3-km radius of each point, here taken from the 1-km global DEM, although the model can now be run at finer resolution with the available 90-m SRTM data. This estimate of relief can be shown to be the topographic parameter which is consistent with Equation (2.18.1), for estimating sediment eroded from hillslopes and delivered to the

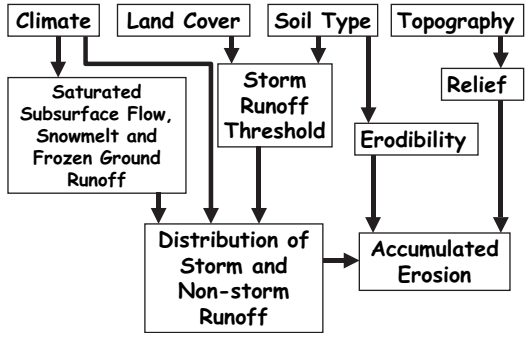

Figure 2.18.6 Component processes included in the PESERA/RDI model

channel network. The process model for estimated erosion is finally calculated as the product of three terms, for relief, soil erodibility and the weighted sum of runoff squared. The third term, based on the form of Equation (2.18.1), is a term which integrates the effects of land use and climate.

Any model which operates at a 1-km scale must contain implicit or explicit scale effects. These are thought to be due to the lack of explicit connectivity terms, which lead to neglect of the effects of land-use patchiness and storm burst durations, discussed above. The effect of these has been minimised by calibrating against plot and small catchment scale data sets, so that the values of soil erodibility are linked to the scale of interest. At 1-km resolution, the model shows strongest erosion in southern Europe, and along the margins of mountainous areas, where more erodible materials occur in piedmont areas which still have significant relief. At a finer resolution, the effects of local topography are more evident (Figure 2.18.7), showing enhanced erosion along valley margins.

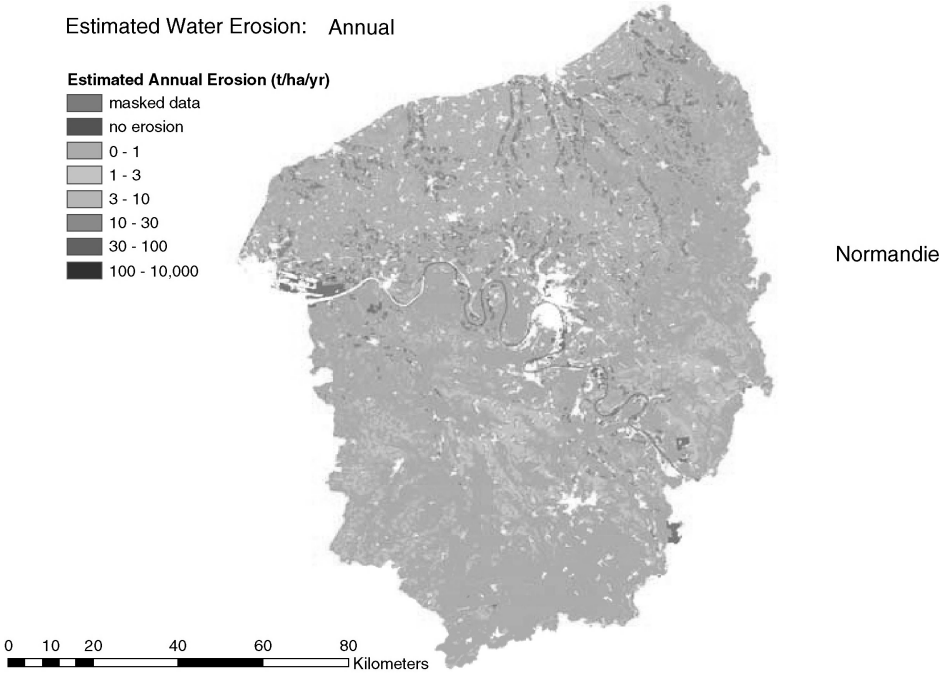

Figure 2.18.7 PESERA/RDI soil erosion risk for Normandy at 50-m resolution

2.18.4 EVIDENCE OF EUROPEAN IMPACTS FROM MODELS AND DATA

If climate scenarios are applied across the whole of Europe within the PESERA/RDI model, based on Hadley Centre scenarios for 2080 (LINK, 2003), but retaining present land use, there is a striking increase in estimated erosion for southern Europe, particularly in winter (Figure 2.18.8). This is thought to be primarily in response to increased temperature and evapo-transpiration, but also to some elements of the more variable changes in scenario rainfall. It is clear that land-use changes will also take place, and that they may accentuate or reduce the impact of the climatic drivers. The expected increases in erosion are also reflected in other modelling work (e.g. Favis-Mortlock and Boardman, 1995), which demonstrated two important points. First, it showed the disproportionate effect of individual large storms on long term totals, and second, the need to take into account positive feedbacks with soil conditions to project the accelerating impact of change over a period of decades.

Land-use changes respond to many factors outside the physical environment, reflecting national and international economics in addition to local and regional policies and support. These are largely outside the scope of the present discussion, but it is worth noting that, within a region, the distribution of land-use change also interacts strongly with local physical factors such as soil type and gradient, even though the overall extent and direction of change is externally driven. The work of Van Rompaey *et al*. (2002) gives one example of this interaction. They use the SEDEM model, in conjunction with Markovian transition matrices based on historical patterns of change, to demonstrate that increases in arable land use in Belgium produce disproportionate increases in erosion because of the expected addition of increasingly marginal land.

Walling *et al*. (2003) compare data on rates of upland reservoir deposition (1–30-km^2 catchment areas) and on downstream flood plain sedimentation in the Ouse and Tweed catchments of Britain. They conclude that increases before and after 1963 in hillslope erosion largely reflect increases in arable land area and lowland drainage, but that flood plain aggradation has been less sensitive to land use change.

Asselman *et al*. (2003) use a suite of models to estimate upslope losses and downstream impacts within the Rhine basin. They combine the RECODES model, based on GAMES and USLE to deliver sediment from upslope areas, with Hadley Centre climate estimates interpolated to 2050, and with a series of independently derived land use scenarios to estimate losses for the next 50 years. They conclude that losses will increase from

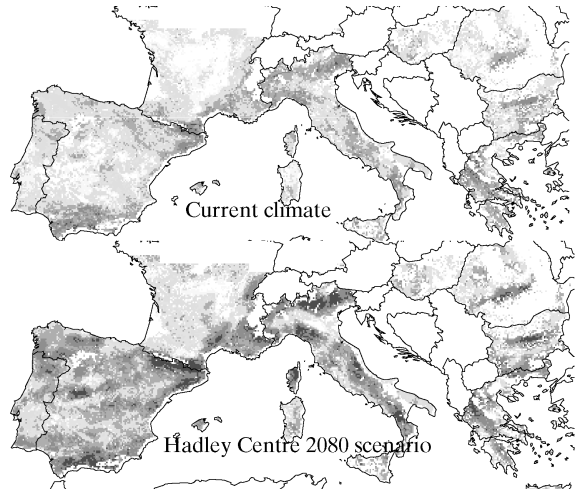

Figure 2.18.8 Comparison of current and 2080 January erosion estimates for southern Europe, based on PESERA/RDI model at 1-km resolution, both with current land use. Shades represent a logarithmic scale of increasing erosion rates (darkest greatest)

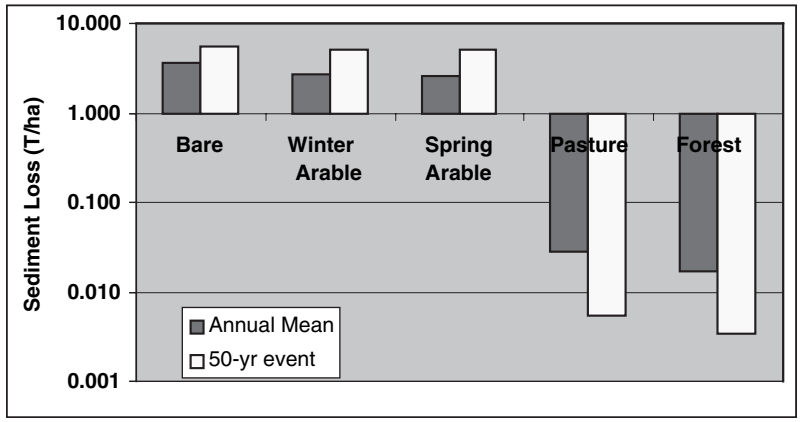

Figure 2.18.9 Comparison of forecasts annual average erosion loss over a run of years and the loss in a single 50-year event for Scotland under different land uses

the Alps, but decrease in the lower parts of the basin, primarily due to reductions in projected arable area as yields increase with warmer climates. However, impacts on the flood plain downstream were strongly buffered and much less evident.

Figure 2.18.9 compares estimated PESERA/RDI annual erosion rates for different land uses. It shows the very large differences associated with different levels of cover, with negligible erosion under forest and pasture, but high rates, associated with high rainfalls, where there is seasonally bare ground during a cultivation cycle. It also shows how extreme events change with cover. Under cultivation, the extreme event, here set at a 50-year recurrence interval, is more erosive than the annual average. This is because the dominant event occurs more frequently than every 50 years. Under this regime, the 50-year and rarer events will be identified with muddy floods which are seen as part of the spectrum of normal erosive events, and justify mitigation through field-scale conservation measures, sedimentation ponds, etc. Under pasture and forest, however, with the dominant event occurring less than once every 50 years, the normal spectrum consists of negligible erosion in most years. Therefore, very occasional extreme events, probably associated with gullying, make a substantial contribution to the long-term average for these types of land cover. For this type of distribution, it is rarely economic to attempt to mitigate erosion, but instead to remediate flood sites, for example by refilling gullies, after an event.

2.18.5 CONCLUSIONS FOR IMPACTS OF ENVIRONMENTAL CHANGE ON SOIL EROSION

The results presented above represent a wider evaluation of the results of process studies, both directly and applied within the context of forecasting models. The conclusions may be summarised as follows. Rainfall increases generally lead to an increase in runoff and erosion. Although, in the semi-arid areas, this increase is offset by an increase in vegetation cover, Mediterranean seasonality always produces a vulnerable period of sparse cover at the end of the summer, so that this vegetation effect (Langbein and Schumm, 1958) is not generally evident in southern Europe. If rainfall is increased through changes in intensity, the impact on erosion is greater than through changes in rainfall frequency. In general, frequency and intensity increase together, but climate scenarios suggest some increase in storminess, so that rainfall effects may be strong. However, integrated scenarios do not show a simple pattern of increase in rainfall, so that the effects of rainfall change remain difficult to forecast reliably.

Temperature increases are widely forecast for the next century, with the greatest increases at high latitudes. Erosion model results suggest that temperature acts most strongly through evapo-transpiration. In northern Europe, increased temperature and evapo-transpiration will tend to reduce soil saturation levels in winter and produce a decrease in erosion, whereas in southern Europe, warming will lead to sparser vegetation and therefore some increase in erosion rates, effectively increasing the area subject to significantly dry summers. Taken together, these changes might favour some concentration on conservation measures around the Mediterranean, associated with a northward migration of the most intensive cultivation, but economic drivers may dictate otherwise.

Responses to land use change have much greater uncertainty owing to the coupling of physical and socio-economic factors. However, all studies show that increases in arable area lead to an increase in erosion. These increases are often disproportionate owing to the use of more marginal land and the cumulative removal of the soil. Therefore, if warmer climates lead to increased crop yields and less land is devoted to arable farming, then erosion will decrease, but if increasing global population and losses of arable productivity in semi-arid areas lead to greater arable cultivation, Europe may instead suffer from increased erosion. The effects on river flood plains are likely to be both more muted and much slower to reveal themselves, so that we should look first at the direction of erosional trends in source areas before making firm predictions.

Erosion is a natural process, and the aim of conservation should never be to eliminate it completely, but rather to return in tolerable levels which are close to natural rates. Erosion is balanced, in the long term, by renewal of both organic matter and mineral soil which maintain soil structure and levels of nitrogen and inorganic nutrients. Excessive erosion changes these balances and, although nutrient levels can be maintained at a cost through fertilisers, the physical loss of eroded material can lead to progressively stonier soils which have a limited capacity for water and nutrients, or to the development of gullies in fine-grained parent materials. Changes in the supply of material eroded from valley sides can also interact with stream channels, leading to the infilling or incision of smaller valleys, with knock-on effects on water supply and flood plain agriculture. Retention of sediment in terraces or valley bottoms, moreover, leaves a legacy for indefinite maintenance and severe incision can rarely be reversed economically.

Where it is considered a priority to minimise the impact of environmental change on erosion, conservation should generally consider the hydrological and sediment balances found under natural conditions. Under a rapidly changing climate, these equilibria cannot be exactly maintained, but our understanding of the balance between natural processes can guide us to a sensitive management of the erosional environment.

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2.19

Muddy Floods

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2.19.1 INTRODUCTION

Water flowing from agricultural fields carrying large quantities of soil as suspended sediment or bedload is referred to as a 'muddy flood'. The process is therefore a fluvial rather than a mass movement one. Muddy flows generally originate on bare or partially vegetated surfaces. The phenomena of muddy floods pre-date the relatively recent adoption of the term.

Early reports of flooding by water from agricultural land describe the erosional forms (rills and gullies) and depositional forms (fans), but say little about water flow and its characteristics (in some cases because it was not observed, e.g. Oakley, 1946). Studies of riverine flooding tend to ignore source areas (probably agricultural fields in many cases).

The term is frequently associated with damage to property, but again this is a relatively recent recognition. In the UK, the link between runoff from agricultural land and property damage was first acknowledged by Morgan (1980) and described in a specific study by Stammers and Boardman (1984). Neither authors used the term 'muddy flood'.

The term 'muddy flood' appears to have been first used in its French form, 'inondations boueuses' by Auzet (1987). Similar terms have been used before in Dutch or French, e.g. 'modderoverlast' in Dutch (mentioned by Schouten *et al.*, 1985) and 'torrents de boues' in French (Papy and Douyer, 1988).

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'An important proportion of the damage related to the erosion is the off-site effect. Here mudflows and muddy floods originating on agricultural fields enter the urbanised areas which are situated in adjacent valley bottoms or at the outlet from usually dry catchments (CEMAGREF, 1986; Papy and Douyer, 1988)' (Auzet *et al*., 1990, p. 383).

Muddy floods seem to be associated with high percentages of arable cultivation and proximity to urban areas (and thus property damage). In this chapter, we limit the use of the term muddy flood to cases where property damage has occurred.

The distinction between muddy flood and mud flow is important geomorphologically since the latter is a mass movement and the former a fluvial event. However, the popular and sometimes the academic literature often confuse the distinction. In France the term mudflow has been used for legal reasons: a mudflow, as a 'natural hazard', enables those affected to claim reimbursement from insurance companies (V Auzet, personal communication).

The term 'muddy flood' (or its synonyms) has been used almost exclusively in relation to events which result in damage to property as a result of runoff from agricultural fields. This undoubtedly introduces a bias in the discussion, since muddy runoff from eroding fields is commonplace. However, we prefer to use the term *sensu strictu,* as it has been used in the literature for the last decade.

2.19.2 RECOGNITION OF MUDDY FLOODING IN EUROPE

In Europe, muddy floods have been described in France, Belgium (Flanders and Wallonia), the UK, The Netherlands (South Limbourg) and Slovakia. Surprisingly, they seem not to have been described in Germany, Italy, Spain, Portugal and Scandinavia. Flash flooding in southern Europe and 'torrents' in some mountainous areas have similarities with muddy floods. The latter are typical of agricultural land whereas flash floods are also typical for more or less 'natural' vegetated areas such as Mediterranean maquis or mattoral. Upland or hilly topography is generally required to generate flash floods. In contrast, muddy floods are typical of lowlands with only minor topographic differences.

There appears to be a close association of muddy floods with ephemeral gullying (Boardman *et al*., 1994), and this suggests wider occurrence of muddy floods, e.g. in Germany, Poland, Portugal, Sweden and Italy – areas where ephemeral gullying has been described. Papy and Douyer (1988) also mention the importance of gullying: 'parce que les ravines, créées par le ruissellement hivernal, peuvent rendre catastrophique un orage de printemps en diminuant le temps de concentration' (gullies developed in the winter season reduce runoff concentration times that can turn springtime storm events into catastrophic inundation events). Steegen *et al*. (2000) also illustrated that gullies developed through the winter season may increase the connectivity between hillslope erosion and sediment delivered to the outlet of a catchment. Boardman (2001a) describes a case from the South Downs, southern England, where gully systems were established as a result of a rainfall event of 90 mm but continued to generate muddy floods throughout the winter (October 2000–April 2001), in some cases with daily rainfall events as low as 4 mm day^{-1}.

The lack of awareness of muddy floods in some areas may be because of lack of damage to property by runoff because urban or village areas are not located close to vulnerable arable land. It is also likely to be due to lack of reporting.

One reason for lack of awareness has been the concentration of erosion studies on experimental plots of limited size. Ephemeral gullying and associated processes (muddy floods) cannot be investigated at this scale. De Ploey's (1989) recognition of erosional systems, including ephemeral gullying, at the catchment scale was an important step forward.

Very partial reviews of muddy flooding in the UK (due to lack of national data) were presented by Boardman (1994, 2000, 2001b). More complete reviews of muddy floods on the South Downs were given by

Boardman (1995a, 2003) and Boardman *et al*. (2003). Successful curtailment of muddy flooding was discussed by Evans and Boardman (2003). Muddy flooding in central Belgium, the Pays de Caux, South Downs and South Limbourg was reviewed by Boardman *et al*. (1994). Similarly, muddy flooding in central Belgium was reviewed by Verstraeten and Poesen (1999) and Bielders *et al*. (2003). Schouten *et al*. (1985) presented an overview on muddy flooding in South Limbourg, The Netherlands. Contributors to this volume provide updates of muddy flooding in South Limbourg (Chapter 1.31), France (Chapter 1.29) and western Slovakia (Stankoviansky 2002, 2003; Chapter 1.11).

2.19.3 THE CHARACTER OF MUDDY FLOODS: SIZE, FREQUENCY, DENSITY: SOME REGIONAL COMPARISONS

There is limited information about the size, frequency and density of muddy flooding. Most of this comes from Belgium, the UK, South Limbourg (The Netherlands), France and Slovakia.

2.19.3.1 Belgium

2.19.3.1.1 Flanders

In the hilly regions of southern Flanders, characterised by highly erodible loess-derived luvisols, some details are available from three events.

As a result of a muddy flood at Huldenberg in May 1997, 15 t of sediment had to be cleared from the roads in the village. In total, 30 man-working days were needed for the cleaning. According to the technical staff, this was a normal event that takes place every year. In June that year, more muddy floods hit the village but no information is available on the other events.

At Hoegaarden in May 1999 a severe rainstorm event (70 mm in 1 h) resulted in an enormous muddy flood event (Figure 2.19.1). A small flood retention pond was completely filled with 700 t (35 t ha^{-1} in a catchment of 20 ha). The pond was not big enough for this event so it overflowed and the village was flooded. In the streets of Hoegaarden, a blanket of mud was deposited ranging in thickness from 1 cm to more than 30 cm locally. An educated guess is that between 200 and 500 t of sediment were deposited on about 2 km of streets. Such severe muddy floods occur every 1–2 years in Flanders but (almost) always in different villages, for instance, Heks-Vlijtingen in June 1996 (Takken *et al*., 1999) and Velm-Gingelom in May 2000. All these villages are about 30–50 km from each other.

The village of Bertem was hit by four muddy flood events between 20 May and 14 June 1997. No figures on sediment volumes are available. However, local staff had to work for 515 h to clean up all the mud and repair some infrastructural damage.

On the basis of newspaper reports for the period 1987–97, the number of muddy floods within the area surrounding Leuven (1100 km^2) was estimated at 0.5 per 100 km^2 yr^{-1}. This figure will be an underestimation since not all muddy floods are reported in the newspaper, and because it was not always mentioned whether a flood event had a muddy character or not. The total number of small-scale flooding events (including the muddy ones) is 3 per 100 km^2 yr^{-1}.

In the southeastern part of Flanders (in the southern part of the province of Limburg), every location affected by muddy floods was recorded. In this area of 408 km^2, 48 locations suffered from muddy floods, which in combination with a frequency distribution of reported events, yields an average number of locations hit by muddy floods of 4.5 per 100 km^2 yr^{-1}. This figure will be an overestimation keeping in mind that during a single muddy flood event, several locations are hit simultaneously (Verstraeten and Poesen, 1999).

Figure 2.19.1 Photograph of mud on the roads in Hoegaarden (with kind permission of Gert Verstraeten)

A conservative estimate of the mean value of muddy flood incidence for the whole affected area in southern Flanders will therefore probably range from 1–3 per 100 km^2 yr^{-1}.

Muddy floods are mostly limited to the period May–June when the dominant summer crops such as maize, sugar beet and potatoes have a low vegetative cover and when the first intense convective rainstorms occur (Verstraeten and Poesen, 1999).

2.19.3.1.2 Wallonia

Wallonia covers an area of 16 800 km^2 in southern Belgium. During the period 1991–2001, 50 % of the 262 municipalities of this region were affected by flood events resulting directly from runoff originating on agricultural land. Agricultural runoff floods (ARFs), which are often muddy by nature, are most common in the loess belt region, which occupies approximately 45 % of the Walloon territory (Figure 2.19.2). Some 66 % of the municipalities in the loess belt region were affected by agricultural runoff floods during the 1991–2001 period, with flood-related damage being rated medium to high by 67 % of the flood-affected municipalities. This is in contrast to the remainder of the Walloon region where only 28 % of municipalities suffered ARFs during the same period. This difference can be attributed to the silt-loam to sandy loam texture of the topsoil in the loess belt region, which renders the soils prone to sealing and favours runoff, and to the much higher percentage of cropland in the loess belt (46 %) compared with the remainder of Wallonia (6 %).

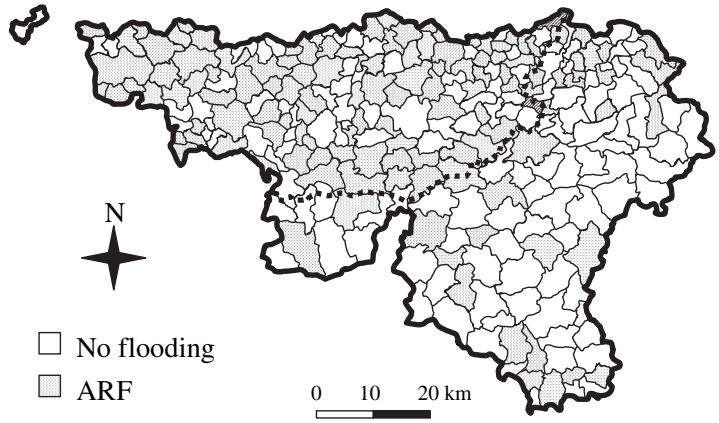

Figure 2.19.2 Map of municipalities affected by agricultural runoff floods (ARF) in Wallonia (1991–2001). The dashed line marks the approximate southern boundary of the loess belt region. (Adapted from Bielders *et al*., 2003)

2.19.3.2 The Netherlands (South Limbourg)

The Landinrichtingsdienst (1983) published a first inventory of 153 flooding locations in South Limbourg. Schouten *et al*. (1985) gave a first account of the extent, spatial distribution, rate, causes, damage and control of erosion, and Van der Helm and Schouten (1986) presented a detailed inventory of 600 erosion sites. Schouten *et al*. (1985) drew attention to the off-site costs of soil erosion. Kwaad *et al*. (Chapter 1.31) review in some detail erosion and soil conservation in this region.

2.19.3.3 United Kingdom

Boardman *et al*. (2003) report 138 incidents of property damage by muddy floods on the eastern South Downs in the years 1976–2001. At several sites flooding occurred in more than one year and at some there were multiply cases of flooding in the same year.

2.19.3.4 France

According to the French database, the total number of muddy floods or 'coulées boueuses' reported in the period 1985–95 equals 5812 (IFEN, 1998). For the two most seriously 'hit' regions, Nord-Pas-de-Calais and Picardie, the total number of muddy floods equals 921, which corresponds to an average density of 2.8 per 100 km^2 for the 10-year period or 0.28 per 100 $km^2\,yr^{-1}$.

2.19.3.5 General

Comparison across European regions is hampered by lack of data. However, the recently published map of mud flood density in France (1985–95) is instructive (Le Bissonnais *et al*., 2002). In the northern Paris Basin densities are >0.7 and in some areas >2.6 events km^{-2}. In the South Downs the comparable value is 0.4 events km^{-2}, but over a 25-year period (1976–2001). The northern French landscape is more susceptible to muddy floods, having large, open fields, more erodible stone-free loess soils and spring-planted crops such as maize, potatoes and sugar beet in addition to winter cereals – there is therefore a risk of runoff in both the

autumn and spring/summer, the latter generated by convectional storms. The French database also contains some cases of riverine flooding (Y Le Bissonnais, personal communication). These factors probably explain the greater density of muddy floods in northern France. Comparable values for the Leuven region and southern Limburg in Flanders are 1–3 events per 100 $km^2\ yr^{-1}$ or 0.1–0.3 events km^{-2} for a 10-year period or 0.25–0.75 events km^{-2} for a 25-year period. For Flanders, only muddy floods that result in property damage are included in these figures.

There are few data relating to flow and sediment concentration in muddy floods. Shallow flow in a valley-floor gully was sampled at Rottingdean, South Downs, on 17 October 1987. Velocities of around 1 m s^{-1} were measured in a flow 2.0 m wide × 0.04 m deep, but this is to be compared with flow at the same site during heavy rain a few days previously of 50 × 1 m. At Breaky Bottom, South Downs (15 October 1987), sediment concentration of 2.3 g l^{-1} was recorded 30 min after rainfall had ceased. These are much lower values than in valley side rills (Table 3 in Boardman, 1988). Muddy floods at Breaky Bottom in 1987 resulted in 35 h of flooding of a vineyard and farmhouse, with an estimated transport of 635 t of soil to the site. This compares with a loss of ca 3440 t from rills and gullies in the catchment.

2.19.4 CAUSES OF MUDDY FLOODS

The primary cause is the existence of large areas of arable land in close proximity to urban areas or rural villages (occasionally isolated farmhouses are affected). Areas of highly erodible loess-derived soils in western Europe also seem susceptible to muddy floods in association with the frequent occurrence of concentrated flow in valley bottoms and the development of ephemeral gullies (e.g. de Ploey, 1989). The relationship between the aspects of the physical landscape (including meteorology) and vulnerable property giving rise to risk or damage is shown in Figure 2.19.3.

For the loess belt region in Wallonia, a strong positive correlation exists between the probability of occurrence of ARFs at least once in 10 years and total agricultural land (cropland + grassland) per municipality, which supports the assertion that ARFs originate from agricultural land and highlights the fact that the occurrence of ARFs in a given municipality is first and foremost a function of the total area of agricultural land that can serve as a source of runoff. At the scale of Wallonia, land use plays an important role in explaining the occurrence of ARFs. Comparing villages with a high frequency of ARFs (at least once every other year on average) with villages not affected by ARFs, it appears that the area of cropland per municipality is positively correlated with the occurrence of ARFs ($p < 0.001$), whereas there is a negative correlation between the occurrence of ARFs and the areas of forest ($p = 0.03$) and grassland ($p = 0.05$). Furthermore, the probability of the occurrence of ARFs with a high frequency increases faster for row crops than for cereals (Figure 2.19.4). This is likely related to the higher soil cover provided by autumn-drilled cereals (wheat and barley) in spring and early summer when the most erosive rainstorms occur (Laurent and Bollinne, 1976). At the scale of the loess belt region, no significant correlation could be found between land use and muddy flood occurrence. This is because land use in this region is much more homogeneous but also because the geographic extent of muddy floods is often restricted to small areas within a municipality and therefore average land use does not necessarily properly reflect the source of the flood water. In addition, factors other than land use may contribute to the occurrence of ARFs, including extreme climatic events, topography or the extent and position of public or private property with respect to the source of runoff and sediment (Boardman *et al.*, 1994). Localized ARFs may go unnoticed if no damage is associated.

In a normally dry valley network, Evans and Boardman (2003) showed that if erosion affected more than 30 % of the basin, the runoff flowing from the fields was likely to cause downvalley flooding. An increase from 30 to 40 % of the catchment affected by erosion and runoff greatly increased the continuity of flow down the valley from 3.1 to 6.5 km of ephemeral gullies.

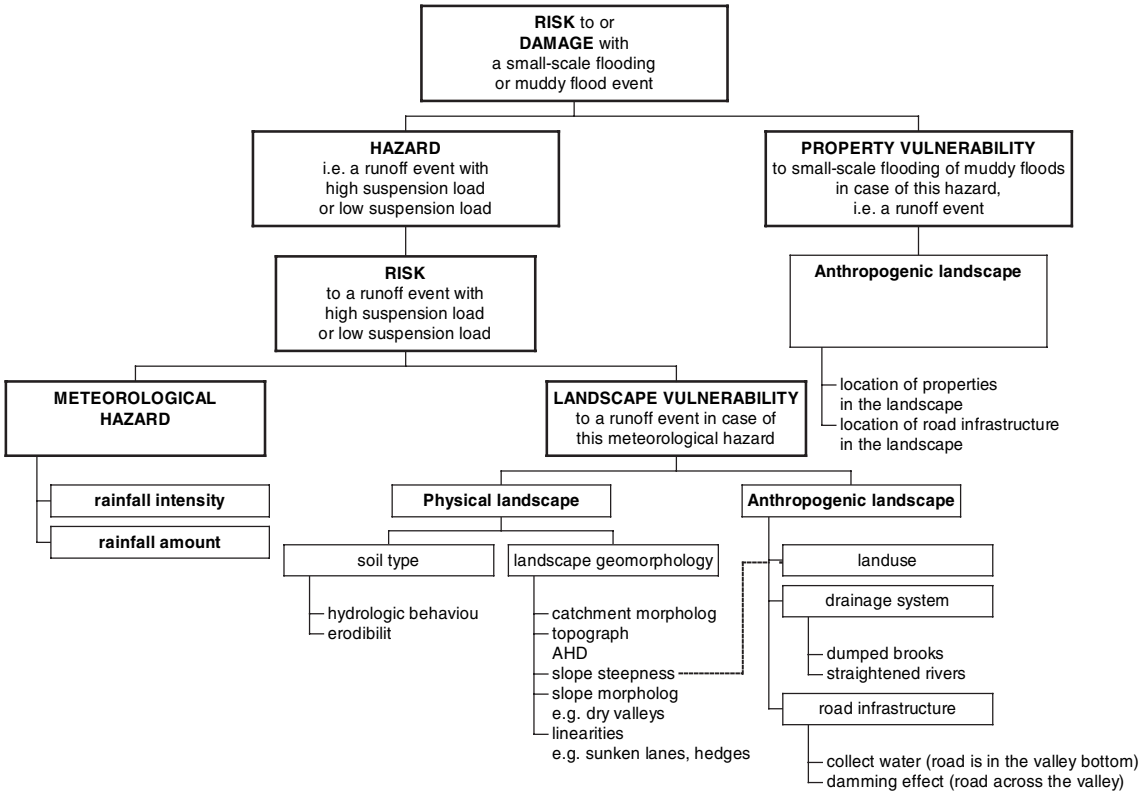

Figure 2.19.3 A structural model illustrating the factors influencing the nature of small-scale flooding and muddy floods in central Belgium (AHD = absolute height difference) (Verstraeten and Poesen, 1999, with kind permission from Elsevier)

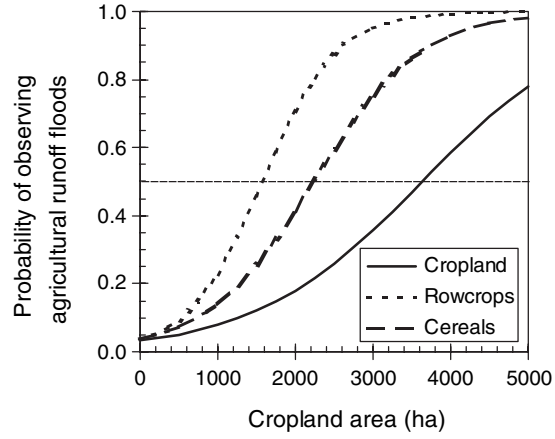

Figure 2.19.4 Probability of occurrence of agricultural runoff floods (ARF) at least once every other year in Wallonia as a function of the area of cropland, cereals and row crops. Only municipalities affected solely by ARFs were considered. The curves indicate that, on average, at least one municipality out of two (50% probability) will experience ARFs at least every other year when the areas of cropland, cereals and row crops exceed 3600, 2200 and 1600 ha, respectively

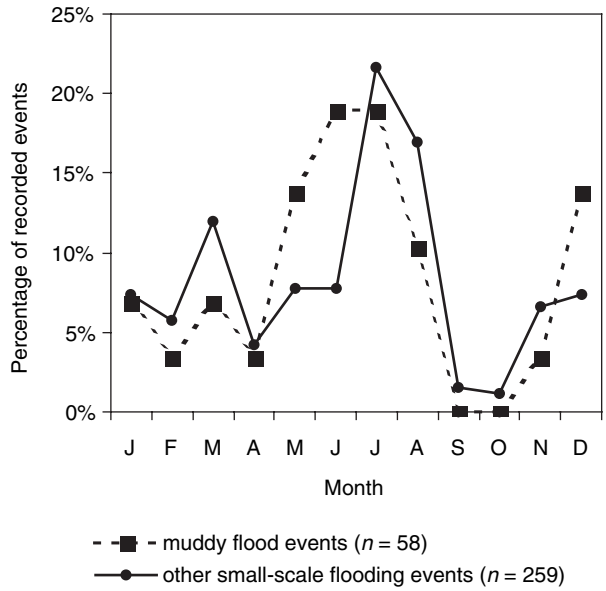

Figure 2.19.5 Intra-annual distribution of reported muddy flood events and other small-scale flooding events in the Leuven area for the period January 1987–July 1997 based on newspaper reports (after Verstraeten and Poesen, 1999, with kind permission from Elsevier)

Issues of increased connectivity on spatially extensive arable surfaces are also relevant to areas where farmland has been reorganised into large-scale blocks of similar land use. Most notably, the process of collectivization under communist regimes in eastern Europe seems to have increased the risk of muddy flooding (Chapters 1.10 and 1.11). To some extent, these social and agricultural changes mirror the intensification of agriculture in western Europe.

The seasonality of muddy flooding is well recognised. It is based on the occurrence of high-intensity, convective rainfall events, which tend in Europe to be summer phenomena (e.g. Boardman *et al*., 1996), and the existence of bare ground. In many areas, summer flooding is associated with land uses such as maize, sugar beet and potatoes, whereas autumn and winter floods are related to autumn-drilled cereals and cyclonic rainfall.

In the Leuven area, most small-scale flooding occurred in the May–August period with a secondary peak in November–March (Verstraeten and Poesen, 1999). Muddy floods show a slightly earlier summer peak compared with all small-scale flooding (Figure 2.19.5). Similar seasonal patterns are reported from northern France (Papy and Douyer 1991). In the South Downs, southern England, the distribution is more markedly an autumn–winter one because of the dominance of winter cereals in the landscape (Boardman *et al*., 2003).

2.19.5 IMPACTS AND COSTS OF MUDDY FLOODING

Virtually all off-farm costs of erosion in UK are associated with ephemeral gullies and muddy floods (rather than rilling). This is because ephemeral gullies are the result of large quantities of water being concentrated in valley bottoms where housing may be located. However, Poesen *et al*. (2003, Figure 6) show that for larger storms the contribution of gullies to sediment production declines. Hence it may be that gullies are chiefly

TABLE 2.19.1 Private costs for selected small-scale flooding events (updated from Verstraeten and Poesen, 1999)

Municipality	Year of damage to property	Cost to private persons at year of damage (BEF)	Actual cost (2004) (€)[a]
Bertem	1978	747 183	65 859
Bertem	1981	111 300	8 475
Bertem	1982	2 796 074	202 758
Bertem	1983	1 197 112	82 675
Bertem	1986	6 000 000	357 951
Bertem	1986	6 000 000	357 951
Bertem	1986	10 700 000	638 346
Bertem	1992	1 197 112	53 293
Overijse	1980	100 000	7 995
Overijse	1985	750 000	46 981
Holsbeek	1995	1 845 792	70 983

[a]Using a mean discount rate = 5 %; €1 = 40.3399 BEF.

important as a form of efficient linkage for water and sediment between slopes and the site of damage (see also Takken *et al*., 1999, and Steegen *et al*., 2000).

Muddy floods cause serious financial damage to people and to public infrastructure. For a number of events in Flemish municipalities, damage costs are available (e.g. Table 2.19.1). The mean annual cost to private property at the level of a municipality varies between 10 and €30 ha^{-1} (actualized figures for 2004). A rough estimation of the total annual cost of muddy floods to private properties in the affected areas in Flanders (5500 km^2) thus equals €5.5–16.5 $\times 10^6$. Financial damage to public infrastructure (clogging of sewers, cleaning roads, dredging drainage ditches, etc.) is of the same order of magnitude. The cost will depend on the severity of the soil loss and also the vulnerability of the housing properties, including housing density in critical areas (see also Figure 2.19.3). Since this varies considerably between Flanders and Wallonia, no extrapolation of total costs to the whole of Belgium can be made. Finally, the psychological damage to people that have to face muddy floods on a regular basis should be stressed.

For South Limburg (The Netherlands), Schouten *et al*. (1985) reported mean annual damage figures for several municipalities (Table 2.19.2). The total figure of €92 521 corresponds to €5.5 ha^{-1}. These figures include almost all the costs related to damage on public infrastructure, but, not all costs to private properties are included because no official registration exists. Since this was done for the values listed in the table for Belgium, it could partly explain the difference between the per hectare value for Belgium and South Limbourg, The Netherlands.

Some costs resulting from muddy flooding on the South Downs were given by Boardman (2003).

In France, the cost of muddy flooding is extremely high (Chapter 1.29). Muddy flow records from 1985 to 1995 yield 5579 catastrophic events (Le Bissonnais *et al*., 2002).

2.19.6 PREVENTION OF MUDDY FLOODING

Papy and Souchere (1993) noted that local communities cannot be effectively protected from muddy flooding by the construction of 'storm basins': the solution to the problem lies in changes in cultivation and spatial planning upslope.

In South Limbourg, a series of policy and farm-level measures have been taken since the recognition of the problem of muddy flooding in the 1980s. Farmers accept some responsibility and aim to provide a level of

TABLE 2.19.2 Mean annual costs to public and private infrastructure for seven municipalities in South Limbourg and for the local Water Board (original data from Schouten *et al*., 1985)

	Original data from 1985 (Fl) (Dutch guilders)			Actual costs (2004) (€)[a]		
Municipality	Public infrastructure	Private properties	Total	Public infrastructure	Private properties	Total
Gulpen	240 000	20 000	260 000	15 034	1 253	16 287
Nuth	17 000	30 000	47 000	1 065	1 879	2 944
Onderbanken	10 000		10 000	626	0	626
Schinnen	4 000		4 000	251	0	251
Valkenburg	370 000	200 000	570 000	2 3177	12 528	35 706
Voerendaal	85 000	200 000	285 000	5 325	12 528	17 853
Wittem	191 000	10 000	201 000	11 965	626	12 591
Waterboard	100 000		100 000	6 264	0	6 264
Total	1 017 000	460 000	1 477 000	63 706	28 815	92 521

[a]Actualised figures using a mean annual discount rate of 5%. 1 Fl = €0.4538.

protection for communities equal to the level provided by small grains, e.g. winter wheat. The most recent Conservation Ordinance (2003) lists measures that farmers are obliged to follow but offers the alternative of developing a farm conservation plan using a scoring system for different measures. Specific measures are required for problem areas where erosion and damage occur and these are developed in concert with the Water Board and municipalities (Chapter 1.31).

In Belgium (both Flanders and Wallonia), the solution to muddy floods has long been sought in the construction of flood retention ponds. In Flanders alone, more than 200 retention ponds have been constructed in recent decades. This strong reliance on retention dams has recently changed (Verstraeten *et al*., 2003). Nowadays, also soil conservation measures such as grass buffer strips, set-aside, afforestation and also grassed waterways, no-till, reduced tillage and strategic grasslands are being promoted to farmers. A recent approach in the soil conservation policy in Flanders is close interaction between the administration, farmers and scientists through discussion forums and demonstration projects. For more information on soil conservation policy in Flanders, see Chapter 2.23 and Verstraeten *et al*. (2003).

As in Flanders, the construction of retention ponds has long been favoured by the Walloon administration for the control of muddy floods. Problems associated with these structures have been discussed elsewhere (Verstraeten *et al*., 2003). Because of problems related to the maintenance of retention ponds, the cost of construction and the space they occupy given that many ponds fully serve their purpose only once or twice in tens of years, a new approach is currently being considered in Wallonia for areas where muddy floods are not too frequent: temporary immersion zones. Temporary immersion zones are spots in the landscape where runoff water naturally converges. Small retention dams equipped with a sluice are constructed at these locations. In case of high runoff risk, the sluice is closed and water is slowly released over a period of a few days. Land use at these locations is unaltered. Only in case of flooding does the landowner receive financial compensation for any damage that may have occurred (damage to crop or grassland, for instance). In addition to the construction of retention ponds and temporary immersion zones, field-level measures are being promoted within the framework of the EU agri-environmental scheme. These include grass buffer strips at field edges that help infiltrate runoff water and filter sediment and associated pollutants, and also winter cover crops that help maintain a proper soil structure (Bielders *et al*., 2003).

In France, the increase in muddy flooding since the 1970s has led the Ministries of the Environment and Agriculture to support surveys and action in three small regions of the Paris basin and in the Midi-Pyrenees. These actions focus on providing protection with retention basins and promoting better agricultural practices.

Several measures are now available for local and provincial authorities to prevent flooding (Chapter 1.29). For example, since the early 1990s, the village of Erlon in Picardie has adopted an engineering approach to the prevention of muddy floods reaching houses. Field boundaries have been raised and strengthened and ditches dug on upslope sides at points where ephemeral gullies regularly develop. The scheme was designed to detain runoff from events of less than 5-year return period (Chambre d'Agriculture de l'Aisne, undated).

In Upper Normandy, Souchere *et al*. (2003) showed substantial runoff reductions from arable catchments if small areas of grass are strategically located. This finding, based on output from the STREAM model, is very similar to conclusions of Evans and Boardman (2003) for the Sompting catchment on the South Downs.

In the UK generally, there are very few examples of cases where targeted land-use change has been used to solve the problem of muddy flooding. Far more frequently, small dams have been constructed as an emergency response to a flooding problem (Boardman *et al*., 2003). In occasional instances the threat of legal action after flooding has encouraged land owners or managers to make changes in land use to reduce the risk of flooding. The Highways Act has also been used by the Isle of Wight council to encourage landowners to prevent water flow from fields onto highways. Reliance on dams is fraught with risk in that land use change within the catchment or climate change can increase the frequency of large storms (Boardman *et al*., 2003).

In England, the risk of water pollution by runoff from arable fields is now recognised by the Environment Agency. In the Rother Valley, West Sussex, serious sediment pollution of the Rother, a valued trout stream, occurs regularly and in the winter of 2000–01 was extremely serious and widespread (Shepheard, 2003). This has led to the setting up of a pilot Landcare scheme to explore effective changes in farming practice to reduce the risk.

The traditional US approach of grassed waterways has had little success in Europe. A recent alternative suggestion of double drilling in concentrated flow zones may offer a cheap and effective alternative (Gyssels *et al*., 2002).

2.19.7 CONCLUSION

To some extent, muddy floods are now recognised as a discrete geomorphological category. They are fluvial phenomena, not mass movements. They are associated with bare ground which in north, central and eastern Europe is most likely to be the product of arable farming (they are therefore a seasonal event). The term 'muddy flood' is best confined to situations where flooding of human structures occurs or natural resources such as fish stocks. They can be described in terms of risk. The risk is related to coincidence of bare ground, vulnerable soils, topography and rainfall.

Muddy floods are closely associated with ephemeral gullying but have been less well studied. In the future, there is likely to be an increase in the risk of muddy flooding in areas where land use remains the same or arable uses expand and also if property is built in vulnerable locations. This is especially the case where predicted climate change will increase the frequency of extreme events (e.g. Hulme *et al*., 2002). However, the risk may decrease as a result of protective measures such as the construction of retention ponds or agri-environmental measures which encourage beneficial land-use change.

Traditionally, in areas affected by muddy floods, the only option that was generally considered was to construct dams and ponds in order to protect communities. During the last decade, other options based on changes in land use and farming practice have been introduced, especially in South Limbourg, Flanders, northern France and Wallonia.

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Off-site Impacts and Responses

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2.20

Reservoir and Pond Sedimentation in Europe

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2.20.1 INTRODUCTION

One of the most important off-site consequences of soil erosion is the deposition of sediment within reservoirs and ponds. This reduces storage capacity and, consequently, also their useful lifetime. It is estimated that the annual loss in storage capacity of the world's reservoirs due to sediment deposition is around 0.5–1%, and for individual reservoirs these values can be as high as 4–5 % (WCD, 2000). These high rates of storage loss pose a serious threat to the economic sustainability of the reservoir that needs consideration. A detailed inventory on reservoir sedimentation rates exists for the USA (Dendy *et al*., 1973; Dendy and Champion, 1978; Renwick, 1996), but this is lacking for most European countries. Nevertheless, also in Europe, many dams exist and many of the reservoirs face serious sedimentation problems. A report of the World Commission on Dams (WCD, 2000) showed that, at present, there are 5480 large reservoirs in Europe (excluding Russia), with a total capacity of no less than 383.6×10^3 hm^3. The most important reservoir nations in Europe are Spain, France, Italy, the UK, Norway, Albania and Romania. In addition to the larger reservoirs, thousands of smaller

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reservoirs, flood control ponds, irrigation ponds, farm ponds and other man-made water bodies exist throughout Europe. However, no detailed inventory on those smaller sized reservoirs exists at all. For the USA alone, it is estimated that there are at least 2.6 million small, artificial water bodies (Smith *et al*., 2002), or a density of 0.33 km^{-2}. Although the total density of such small reservoirs may be less in Europe, the total number of such small artificial water bodies can easily be as high as 0.5–1 million (densities of 0.1–0.2 km^{-2}).

All these dams and ponds experience sediment deposition (Figures 2.20.1 and 2.20.2); however, the intensity of the sediment deposition process varies tremendously from one reservoir to the other. Combined, reservoir and pond sedimentation can have an important impact on the sediment dynamics of many river basins in Europe. For the River Ebro, it is estimated that the actual sediment delivery to the delta is only 1 % of the value before the dam construction era (Guillen and Palanques, 1992). As a consequence, marine processes erode the delta, whereas salt seawater intrudes the area. Reduced sediment delivery to larger rivers may also result in river channel incision with various economic and environmental consequences, such as undercutting of infrastructure (e.g. bridges), lowering of the groundwater table in the floodplain and increased soil salinization in floodplain areas and bank collapses, which is, for instance, reported for the River Rhône (Arnaud-Fassetta, 2003) and for several Italian rivers (Surian and Rinaldi, 2003).

Although the impacts of reservoir sedimentation have been illustrated for various European rivers, no effort has been made to give an overview on the spatial variability in reservoir sedimentation rates at the European scale. Although reservoir sedimentation data are a widely used technique to derive sediment yield values for the upstream catchment, such data are very rare. There are several reasons why data on reservoir sedimentation and sediment yield data are hard to find for Europe. First, different types of reservoirs and ponds exist for various reasons (drinking water, power generation, floodwater protection), implying that these are managed by diverse companies and government agencies at local or national levels. For most of these managers,

Figure 2.20.1 Sediment deposition in the large La Fuensanta reservoir, SE Spain. According to Avendaño Salas *et al*. (1997), this reservoir has a storage capacity of 235 hm^3 and experiences a mean annual storage capacity loss of about 0.2 %. Drainage area equals 979 km^2. (Copyright Gert Verstraeten, with kind permission)

Figure 2.20.2 Sediment deposition in a small flood retention pond in Belgium. It has a storage capacity of 10 000 m^3 and a mean annual storage capacity loss of nearly 10 %. Drainage area equals 48 km^2 (data from Verstraeten, 2000). (Copyright Gert Verstraeten, with kind permission)

sedimentation may be problematic but not important enough to measure it accurately. If it is measured, often only sediment volumes are reported, which is important for managing the reservoir, but not enough for calculating sediment yield. The lack of data is even more pronounced for smaller ponds operated by government agencies. In some cases, especially for private companies, high sedimentation rates are being kept secret as confidential information. Finally, the lack of a single European agency that coordinates all information on soil erosion and reservoir sedimentation, comparable to the National Resources Conservation Service (NRCS) in the USA with the RESIS database (Reservoir Sedimentation Information System), hampers a better understanding of reservoir sedimentation in Europe. This paper brings together sedimentation data for nearly 400 European reservoirs and ponds from various sources. This dataset was used to calculate and analyse the spatial variability of sediment delivery in various European countries. Furthermore, the impact of man-made reservoirs and ponds on sediment delivery at the continental scale is discussed.

2.20.2 SEDIMENTATION RATES IN EUROPEAN RESERVOIRS AND PONDS

Data used in this analysis were taken from sources reported in Table 2.20.1. Table 2.20.2 provides generalized data on sedimentation rates for 392 reservoirs and ponds in a number of European countries. Mean annual volumetric sedimentation rates for all studied reservoirs and ponds are shown in Figure 2.20.3. It should be noted, however, that the size and type of the reservoirs and ponds studied differ greatly from country to country, making it difficult to compare the results. For instance, for Belgium, only data for very small flood-control ponds are analysed, whereas for Spain only larger reservoirs have been studied (see also Figures 2.20.1 and 2.20.2).

TABLE 2.20.1 Data sources on reservoir and pond sedimentation used in this study

Country	Data source
Austria	Cyberski, 1973; Tschada and Hofer, 1990
Belgium	Verstraeten and Poesen, 2001a; Verstraeten, 2000
Czech Republic	Krasa *et al*., 2005; Van Rompaey *et al*., 2003a; Cyberski, 1973
France	Cadillon *et al*., 1981; Combes, 1981; Fauroux, 1981; Vivian and Thomas, 1982; Cravero and Guichon, 1989; Blanc *et al*., 1989; Brochot, 1993
Germany	Baade, 2001. Cyberski, 1973; Schröder and Theune, 1984
Italy	Bazzoffi, 1987; Bazzoffi *et al*., 1996; Bazzoffi and Van Rompaey, 2004; Tamburino *et al*., 1989, 1990; Van Rompaey *et al*., 2003b
Poland	Cyberski, 1973; Lajczak, 1996, 2003
Romania	Ionita *et al*., 2000; Rãdoane and Rãdoane, 2005
Slovakia	Janský, 1992; Cyberski, 1973
Spain	Avendaño Salas *et al*., 1997a,b; Verstraeten *et al*., 2003; Cobo Rayán, personal communication
Switzerland	Beyer Portner, 1998
UK	Al-Jibburi and McManus, 1993; Barlow and Thompson, 2000; Butcher *et al*., 1992, 1993; Charlesworth and Foster, 1993; Curr, 1995; Duck and McManus, 1990; Foster, 1995; Foster *et al*., 2003; Foster and Walling, 1994; Foster and Lees, 1999; Labadz *et al*., 1991; McManus and Duck, 1985; White *et al*., 1996

To overcome this problem partly, we relate the annual loss in storage capacity to the ratio of capacity to catchment area (Figure 2.20.4). For 40 out of the 392 reservoirs and ponds, no storage capacity data are known and, consequently, no annual losses in storage capacity could be calculated. The highest storage capacity losses (>5 %) are found in the smaller ponds of central Belgium, in various medium-sized Alpine reservoirs (especially in south-east France) but also in some larger reservoirs in the Romanian Carpathians. Lowest annual storage capacity losses (<0.1 %) are found for the very large reservoirs with relatively small drainage areas (such as some Swiss reservoirs in the higher Alps, e.g. Grande Dixance) and for the medium-sized reservoirs in the UK.

Table 2.20.3 provides mean and total figures on reservoir sedimentation for the analysed reservoirs. The average annual storage capacity loss for 352 reservoirs is 0.26 %, which corresponds to a total measured sedimentation rate of 62.65 $hm^3\,yr^{-1}$. The total sedimentation rate in all European reservoirs and ponds will be much higher. For Spain alone, Batalla (2002) estimated the total annual sedimentation rate in all large reservoirs at 170 $hm^3\,yr^{-1}$. He used a mean annual storage loss of 0.3 %, a value that corresponds to the median storage loss of the 87 Spanish reservoirs in this study (Table 2.20.1), and applied this to all the large reservoirs in Spain (around 1200). By extrapolating the mean annual storage loss of 0.26 % to all the 5480 large reservoirs in Europe having a total storage capacity of 384×10^3 hm^3 (WCD, 2000), we can assess the total sedimentation rate in large European reservoirs at 997 $hm^3\,yr^{-1}$ (Table 2.20.3). A similar figure of 1074 $hm^3\,yr^{-1}$ was obtained applying the size distribution of observed reservoirs to all 5480 large reservoirs in Europe and using mean observed storage losses for 10 reservoir size classes (Table 2.20.4). The size distribution of the 352 studied reservoirs corresponds more or less to the size distribution of the 5480 reservoirs according to the World Commission on Dams (WCD, 2000).

These figures do not include sedimentation in smaller reservoirs and ponds, which can have the same order of magnitude. For the USA, Smith *et al.* (2002) estimated that half of the sediment storage in water bodies was trapped in large reservoirs and the other half in smaller reservoirs and ponds. For Europe, insufficient

TABLE 2.20.2 Reservoir and catchment characteristics for 392 reservoirs and ponds in 12 European countries

Country	No. of reservoirs with data	Catchment area (km^2)			Capacity (10^3 hm^3)			Annual sedimentation rate (% of storage capacity)			Reservoir sedimentation rate (m^3 km^{-2} yr^{-1})			Total sedimentation rate (hm^3 yr^{-1})
		Median	Min.	Max.	Median	Min.	Max.	Median	Min.	Max.	Median	Min.	Max.	
Belgium	21	8	0.1	48.7	0.008	0.0001	0.025	8.20	2.30	28.60	111	18	955	0.022
Czech Republic and Slovakia	36	13	0.8	412.0	0.106	0.017	0.288	1.76	0.32	9.29	150	4	442	0.086
UK	64	5	0.4	54.4	0.622	0.006	21.4	0.09	0.00	1.76	138	2	1075	0.132
Italy	53	68	0.3	697.0	6.69	0.007	325	0.68	0.00	14.35	497	1	2754	4.441
Spain	87	690	1.3	16952.0	420	1.8	1670	0.30	0.00	4.43	231	1	6581	24.631
Romania	59	542	33.2	19445.0	13.3	0.22	1400	1.49	0.04	13.71	213	6	3208	21.670
France	8	990	3.6	4850	14.7	0.12	1272	3.37	0.90	10.00	240	8	2708	3.482
Switzerland	19	49	16.6	886.9	20.15	0.04	401	0.19	0.01	95.37	475	44	2048	1.749
Austria	7	324	10.0	6250	0.845	0.15	140	10.25	0.01	100	262	37	1851	1.233
Germany	15	30	0.6	1423	–	–	–	–	–	–	35	7	154	0.486
Poland	23	1124	94.0	193900	94.3	8	474.5	0.13	0.01	0.83	129	2	1191	5.155
Total	392		0.1	193900		0.0001	1670		0	100		1	6581	63.087

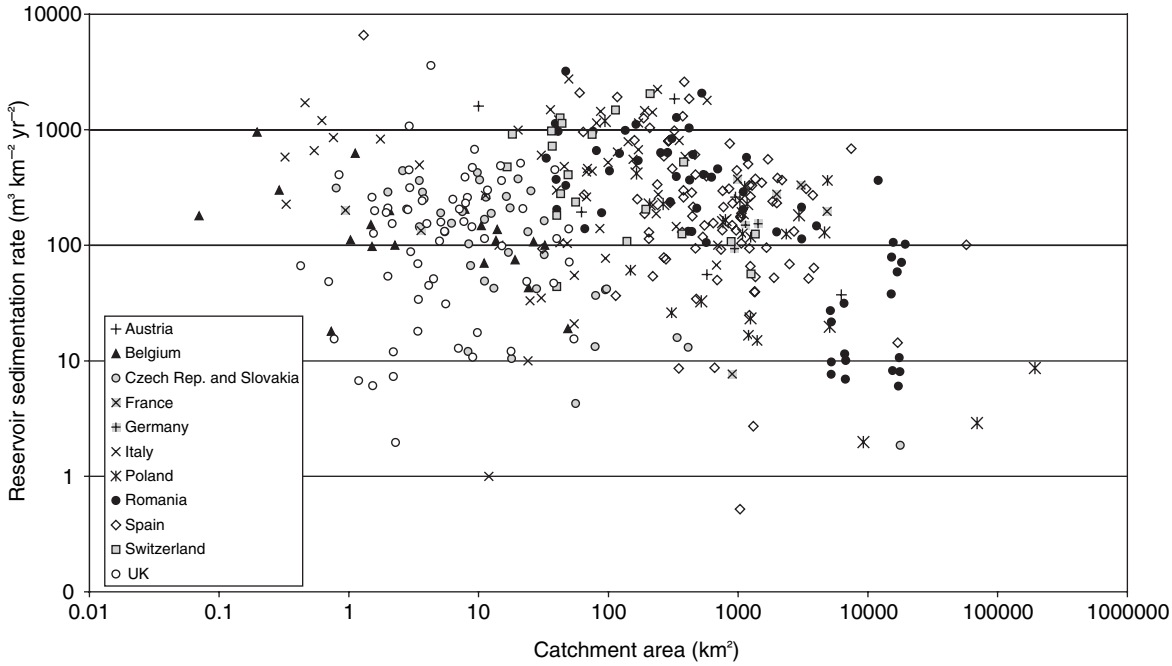

Figure 2.20.3 Mean annual volumetric reservoir sedimentation rates versus catchment area for 392 ponds and reservoirs in 12 European countries

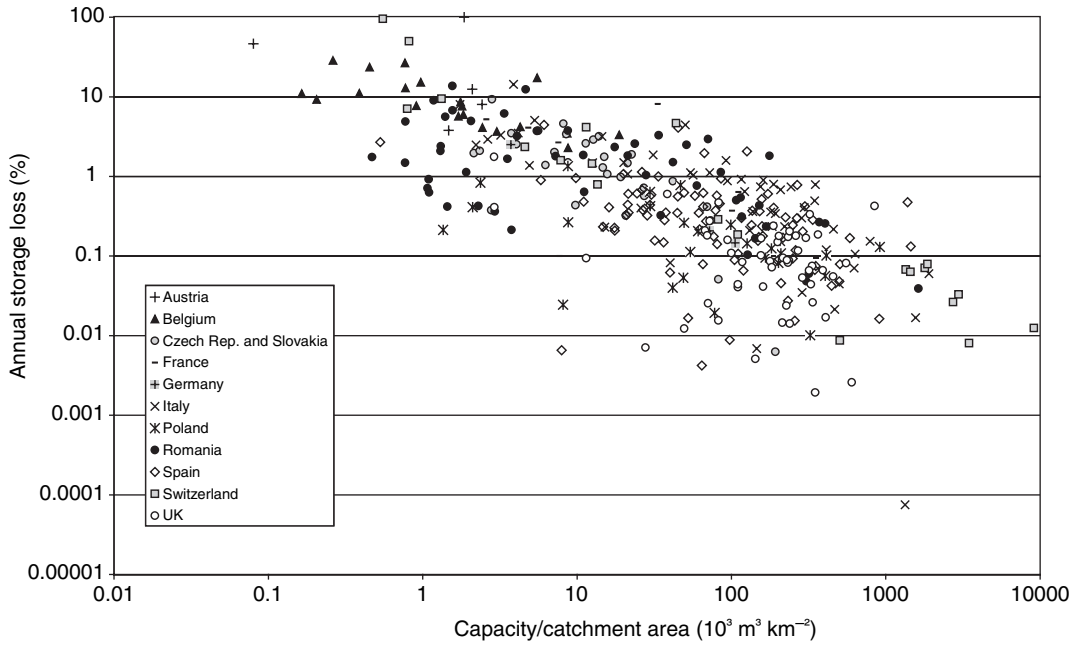

Figure 2.20.4 Annual storage loss versus a capacity to catchment area ratio for 352 ponds and reservoirs in 12 European countries

TABLE 2.20.3 Estimation of total annual sedimentation rate in all 5480 large European reservoirs using mean annual storage capacity loss of 352 studied reservoirs

	This study	Europe
Total No. of reservoirs	352	5480
Total reservoir capacity (hm^3)	24466	383600
Total annual sedimentation volume ($hm^3\,yr^{-1}$)	62.65	997.36
Mean annual storage capacity loss (%)	0.26	0.26

information is available on pond sedimentation rates in various regions. Most of the smaller ponds for which data are available are situated in the loess belt of central Belgium, which is characterized by moderate to high erosion rates (Chapter 1.30). If we assume that their mean annual storage loss of 10 % is representative for the 0.5–1 million ponds in Europe, mean annual sedimentation rates could indeed be as high as 500–1000 $hm^3\,yr^{-1}$. However, if it is assumed that most ponds are situated in regions with lower erosion rates, these values can be much lower, even less than 100 $hm^3\,yr^{-1}$. Total sediment storage in all European reservoirs and ponds therefore probably ranges from 1000 to 2000 hm^3/year. It is clear that smaller ponds can be an important sediment sink in the total sediment budget of large European river systems, but more data on sedimentation rates for smaller ponds are needed to obtain a reliable estimate.

Data for the 5480 large reservoirs are for the whole of Europe excluding the Russian Federation. This corresponds to a total land area of approximately 5.57×10^6 km^2. Hence the mean annual sedimentation rate in larger European reservoirs corresponds to about 180 $m^3\,km^{-2}\,yr^{-1}$ or 1.8 $m^3\,ha^{-1}\,yr^{-1}$. Adding the sedimentation rates in smaller ponds, this value will probably range between 2 and 4 $m^3\,ha^{-1}\,yr^{-1}$.

2.20.3 COMPUTING SEDIMENT YIELD FROM RESERVOIR SEDIMENTATION RATES

Reservoir and pond sedimentation rates are one of the most often used means to assess sediment yield worldwide (Dendy *et al.*, 1973; McManus and Duck, 1985; Van den Wall Bake, 1986; Neil and Mazari, 1993; Foster, 1995; White *et al.* 1996; Verstraeten and Poesen, 2001a). Area-specific sediment yield (*SSY*, $t\,km^{-2}\,yr^{-1}$) can be calculated using

$$SSY = 100\frac{SV \times dBD}{TE \times A} \tag{2.20.1}$$

where *SV* is the annual sedimentation rate in the reservoir of pond ($m^3\,yr^{-1}$), *dBD* the dry sediment bulk density ($t\,m^{-3}$), *TE* the sediment trap efficiency of the reservoir or pond (%) and *A* the catchment area (km^2) draining to the reservoir or pond. The advantage of reservoirs and ponds for the study of sediment yield is that they are abundant, which makes it possible to study sediment yield at a regional scale. Furthermore, reservoirs and ponds also trap bedload sediment. This type of sediment is rarely measured, although it can be very important, certainly in Mediterranean Europe and in mountainous regions.

However, every parameter in Equation (2.20.1) needs to be estimated and this is not always possible, thereby making the sediment yield assessment less accurate (e.g. Salas and Shin, 1999; Evans and Church, 2000; Verstraeten and Poesen, 2002). Many studies that report on reservoir sedimentation rates were not performed with the goal of calculating sediment yield, and therefore not all the necessary information to apply Equation (2.20.1) is known. Especially values for *dBD* and *TE* are rarely provided. Furthermore, if values for

TABLE 2.20.4 Estimation of total sediment deposition values in all large 5480 European reservoirs using size distribution and mean annual storage capacity losses of 352 studied reservoirs

Minimum capacity (hm^3)	Maximum capacity (hm^3)	Median capacity (hm^3)	No. of studied reservoirs and ponds	Estimated number of reservoirs in Europe	Mean annual storage capacity loss for studied reservoirs and ponds (%)	Total measured annual storage loss for studied reservoirs and ponds (hm^3)	Total estimated annual storage loss for studied reservoirs and ponds (hm^3)[b]	Total estimated annual storage loss for all European reservoirs (hm^3)[c]
0	0.01	0.005	13	202	10.07	0.005	0.007	0.10
0.01	0.1	0.05	29	451	4.95	0.05	0.08	1.23
0.1	0.5	0.25	50	778	4.65	0.57	0.70	10.86
0.5	1	0.75	19	296	5.86[a]	0.84	0.84	13.00
1	5	2.5	41	638	1.79	1.93	2.20	34.28
5	10	7.5	30	467	1.42	3.24	3.20	49.74
10	50	25	84	1308	0.99	20.08	24.95	388.40
50	100	75	25	389	0.26	4.53	4.88	75.89
100	500	250	54	841	0.15	18.39	24.30	378.31
500	1000	750	7	109	0.15	13.00	7.88	122.60
Total			352	5480	0.26	62.65	69.01	1074.40

[a]Only 1.73 % if one Alpine reservoir with an annual storage capacity loss of 100 % is left out; this has no major influence on total sedimentation values, however.
[b]Mean annual storage capacity multiplied with number of studied reservoirs and the median capacity (e.g. $13 \times 0.005 \times 0.1007 = 0.007$).
[c]Mean annual storage capacity multiplied with number of estimated reservoirs and the median capacity (e.g. $202 \times 0.005 \times 0.1007 = 0.10$).

TABLE 2.20.5 Statistics on sediment yield data derived from reservoir and pond sedimentation rates

Country	No. of reservoirs with data	Area-specific sediment yield ($t\,ha^{-1}\,yr^{-1}$)		
		Median	Min.	Max.
Belgium	21	2.3	0.4	20.6
Czech Republic and Slovakia	36	2.2	0.1	7.0
UK	71	0.5	0.0	3.8
Italy	53	4.5	0.0	25.1
Spain	65	2.9	0.0	26.8
Romania	59	5.4	0.1	46.8
France	6	4.0	2.8	50.5
Switzerland	19	7.8	0.6	31.4
Germany	12	0.7	0.2	2.0
Poland	18	2.6	0.1	17.2

dBD or *TE* were used to apply Equation (2.20.1), they are often estimated using standard procedures developed for large reservoirs in the USA, but actual measurements are rare (see Verstraeten and Poesen, 2001b, and Verstraeten and Poesen, 2000, for more information on *dBD* and *TE* predictions, respectively). This seriously hampers the use of volumetric sedimentation rates (*SV*) to assess sediment yield.

Statistics on area-specific sediment yield rates in Europe based on 372 reservoirs and ponds are given in Table 2.20.5. In most cases, no reliable measurement of *TE* or *dBD* was made. In those cases, *TE* was estimated using the equation of Brown (1943), which has proven to be useful for UK reservoirs (Butcher *et al*., 1992) and reservoirs in Spain (Avendaño Salas *et al*., 1995). When no information was presented on reservoir capacity, *TE* was set at 100 %, such that a minimal sediment yield was estimated. Estimates for *dBD* were based on *dBD*-measurements of nearby reservoirs with similar sediments. If such information was not available, *dBD* was set at $1\,t\,m^{-3}$.

2.20.4 SPATIAL VARIABILITY OF SEDIMENT YIELD IN EUROPE BASED ON RESERVOIR SEDIMENTATION DATA

Calculated area-specific sediment yields are plotted against catchment area in Figure 2.20.5. A large scatter can be observed, illustrating that for the whole of Europe no significant relation between area-specific sediment yield and catchment area can be established. This large observed variability in area-specific sediment yield for catchments with similar sizes is due to the heterogeneity of the European landscape in terms of climate, topography, lithology and land cover. At the level of individual countries, some significant relations can be drawn, however (Figure 2.20.6). This is the case for Belgium, the Czech Republic and Slovakia, Spain, France, Poland, Romania and Germany. Within Italy, Switzerland and the UK, the variability is still too high to obtain a significant relationship. It should also be realised that part of the variability can also be attributed to errors in the calculation of the *SSY* owing to, for instance, erroneous estimations of *dBD* and/or *TE*. Furthermore, the period for which the *SV* is measured may differ from reservoir to reservoir, which can make a comparison difficult if controlling factors such as climate or land use have changed from one period to another.

Despite these uncertainties, it is evident from Figures 2.20.5 and 2.20.6 that two major regions in Europe can be distinguished with respect to area-specific sediment yield. The first group of countries includes Belgium, Germany, the UK, Czech Republic, Slovakia and part of Italy. A second group includes Romania,

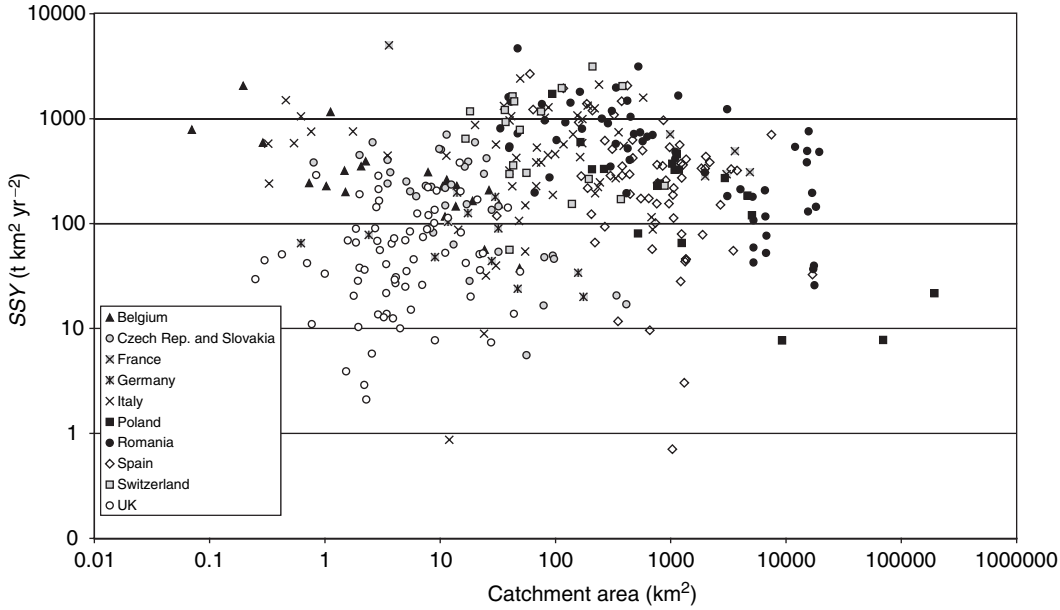

Figure 2.20.5 Area-specific sediment yield (*SSY*, $t\,km^{-2}\,yr^{-1}$) plotted against catchment area (km^2) based on sedimentation rates for 372 reservoirs and ponds in 11 European countries

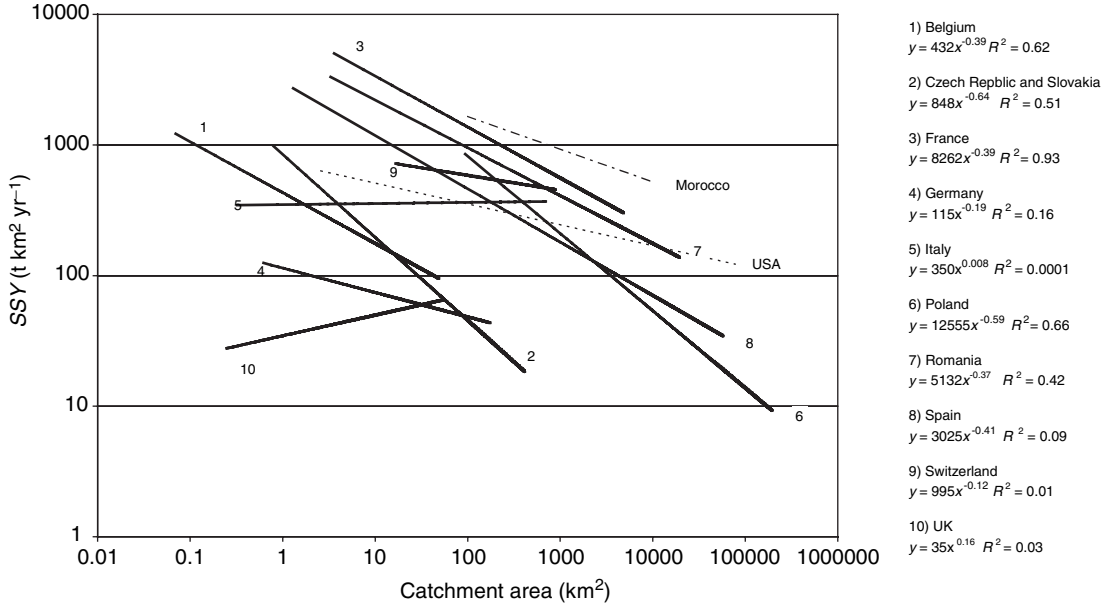

Figure 2.20.6 Relation between catchment area and area-specific sediment yield (*SSY*) for 11 European countries based on reservoir sedimentation rates and compared with similar relations for the USA (Dendy and Bolton, 1976) and Morocco (Lahlou, 1988). Regressions are significant at the 95 % confidence level for all countries except for the UK, Italy and Switzerland

Spain, France, Poland, Switzerland and the other part of Italy. For a given catchment area, area-specific sediment yield values are much lower for the first than the second group of countries.

The first group encompasses regions in the north-western and central European lowlands and tectonically old middle mountains. Topography is less pronounced in these regions with a majority of gentle slopes. In many of the studied areas pasture and forest dominate (e.g. UK and Germany), whereas in other areas cropland is the dominant land use (e.g. Belgium, Slovakia). Highest area-specific sediment yield values in group 1 are observed for catchments in central Belgium with a rolling topography, highly erodible loess-derived luvisols and intensive agriculture. Lowest area-specific sediment yield values in group 1 are observed for forested catchments in the UK Midlands.

The second group are data from regions which are either dominated by mountainous topography (Carpathians in Poland and Romania, south-east Alps in France, Alps and Apennines in Italy, Pyrenees and several Sierras in Spain and the Alps in Switzerland) or that are situated in the Mediterranean region (south-east France, Spain, central and southern Italy, including Sicily), or a combination of the two. Mediterranean regions are generally more prone to erosion and sediment transport owing to the higher rainfall erosivity, sparse vegetation cover and often a highly erodible lithology (e.g. marls). Highest area-specific sediment yield values in group 2 are observed for very steep mountain catchments in Mediterranean France, where the importance of torrential flows is very high, whereas lowest area-specific sediment yield values are seen for the very large catchments in Poland that drain the Carpathians but also encompass large parts of the lowlands at the foot of the Carpathians that do not contribute much sediment. In fact, the larger catchments in Poland should be classified to group 1 and only the catchments situated totally in the Carpathians are part of group 2.

Milliman (2001) made a similar distinction between these two regions when he compared the suspended sediment load data from large rivers in northern and southern Europe. The lowest suspended sediment loads were found for Scandinavian rivers, mainly because of the dominance of older and harder rocks. However, no dataset on reservoir sedimentation rates for Scandinavian countries is available Figure. 2.20.6 also compares the area-specific sediment yield data for Europe with data from the USA and Morocco. It is clear that the median area-specific sediment yield in Europe is more or less comparable to the median trendline observed for 800 reservoirs in the USA (Dendy and Bolton, 1976). The relation for Morocco (Lahlou, 1988) is in accordance with the relations for European Mediterranean countries, which show similar climatic and geomorphological characteristics. The fact that the Moroccan catchments have even more pronounced topography and less vegetation (more arid than Europe) may explain the even higher area-specific sediment yield values than for Spain.

Although there is a clear contrast between Mediterranean regions and north-western and central European regions, variability within every country and region remains high. In Spain, for instance, area-specific sediment yield values for a catchment of $1000\,\mathrm{km}^2$ may vary between less than 1 to more than $1000\,\mathrm{t\,km^{-2}\,yr^{-1}}$. This can be attributed to important regional differences in topography (the Sierras versus relatively flat intramontane basins), geology (marls versus limestone) and the presence of gullies as sediment source but certainly as sediment pathways increasing the connectivity from the hillslopes to the reservoirs (Verstraeten *et al*., 2003; Poesen *et al*., 2003). The highest values for Spain correspond to the trendline for Moroccan reservoirs. These catchments are characterised by the presence of many gullies and low vegetation cover.

For the moment, insufficient data on factors controlling sediment yield (e.g. topography and land use) are available for the corresponding catchments to investigate the spatial variability in a more detailed way.

The high variability in sediment yield is also illustrated in Figure 2.20.7. Although there is a strong significant relation between catchment area and total annual sediment yield, this is only related to catchment area itself (area exponent equals $\sim$1). A huge scatter around the mean trend exists, making it impossible to use this relation to predict the sediment yield in Europe. For instance, there is a 95 % probability that a catchment of $100\,\mathrm{km}^2$ yields a total sediment delivery in between 1000 and 600 000 t.

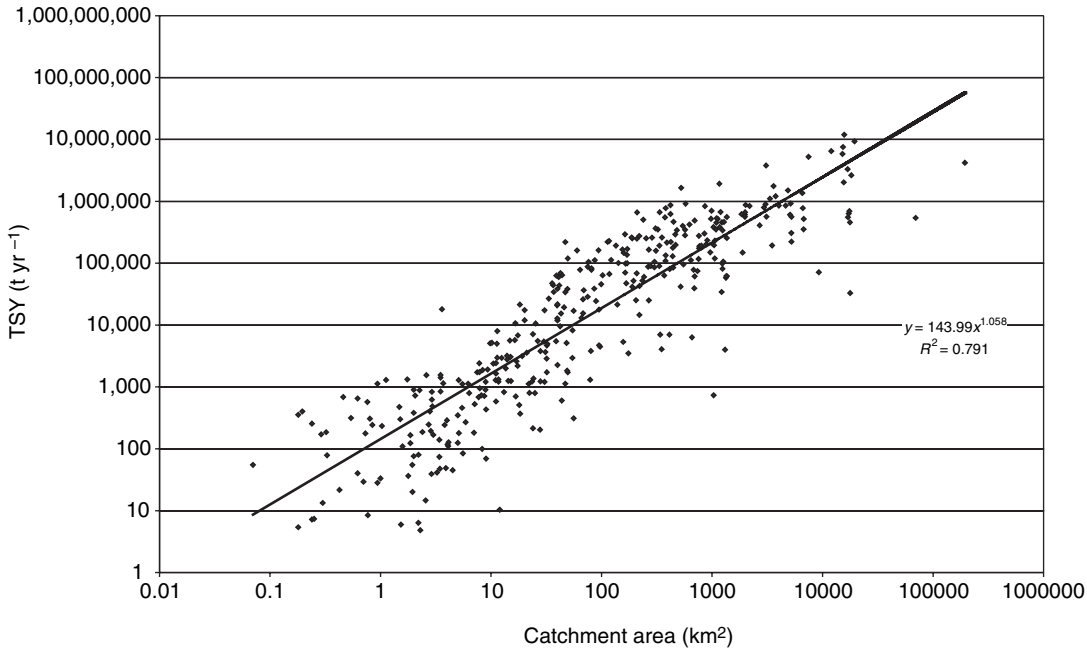

Figure 2.20.7 Total annual sediment yield (*TSY*) versus catchment area for 372 reservoirs and ponds in 11 European countries. Dotted lines represent the 95 % confidence limits for individual predictions of total annual sediment yield

For smaller catchments (and smaller corresponding ponds), there is also a very important temporal variability in sediment yield that makes it difficult to obtain reliable long-term estimates of the sediment yield using sedimentation rates. This is probably best illustrated with sedimentation rates for three small ponds in the Massif des Maures, south-east France. In the first year after a forest fire, sediment deposition rates in small ponds with catchments of 18, 24 and 94 ha reached extreme values of 19.7, 10.6 and 12.0 t ha^{-1} yr^{-1}, respectively. The year after, however, sediment yields dropped to 0.3 t ha^{-1} yr^{-1} in all three catchments (Martin *et al*., 1997). Another example was the complete filling of a small flood-control pond in central Belgium after a local intense storm event. This raised the mean annual area-specific sediment yield over a 3-year measurement period from 8.4 to 20.1 t ha^{-1} yr^{-1} (Verstraeten, 2000).

2.20.5 IMPACT OF RESERVOIRS AND PONDS ON TOTAL SEDIMENT YIELD AT THE EUROPEAN SCALE

It is not possible to estimate the total and mean sediment yield for Europe based on the reservoir data. Especially for Spain and Romania, many dams are located above each other on the same river. Summing the area-specific sediment yield values for reservoirs on the same river would overestimate the true area-specific sediment yield for that river basin. However, a minimum sediment yield can be assessed if only the sediment deposits in the reservoirs are taken into account, and no correction for *TE* is made. For 356 out of the studied reservoirs and ponds, this corresponds to a total value of 56.3 hm^3 yr^{-1} and 70.5 Mt yr^{-1}, thus resulting in a mean *dBD* of 1.25 t m^{-3} (70.5/56.3). Above, we estimated the total annual deposited sediment volume in all

European reservoirs and ponds at 1000–2000 $hm^3\,yr^{-1}$ or 200–400 $m^3\,km^{-3}\,yr^{-1}$. Multiplying this with the mean *dBD* for the studied reservoirs and ponds, a minimum area-specific sediment yield of 250–500 $t\,km^{-2}\,yr^{-1}$ can be established, corresponding to 1250–2500 $Mt\,yr^{-1}$. These are rather high values compared with estimates of total sediment delivery from European rivers to the Atlantic Ocean, the Mediterranean Sea and the Black Sea. Milliman and Meade (1983) estimated this at 230 $Mt\,yr^{-1}$ or 50 $t\,km^{-2}\,yr^{-1}$ (total surface area $4.61\times10^6\,km^2$) based on limited suspended sediment load data for several rivers near their mouth. Although the figures reported by Milliman and Meade (1983) are conservative estimates (they are often based on infrequent suspended sediment sampling that mostly underestimates total sediment load), it is clear that reservoirs and ponds have an enormous impact on the sediment budget for European rivers at the subcontinental scale. All reservoirs and ponds thus trap 5–10 times the amount of sediment that is eventually delivered to the sea. This would correspond to a mean sediment trap efficiency of 80–90 % at the European scale. These estimates are much higher than those given by Vörösmarty *et al*. (2003), who estimated the mean trap efficiency for 88 very large reservoirs (i.e. capacity $>500\,hm^3$) for the whole of Europe at 23 %. They did not estimate the contribution of other large reservoirs [i.e. the 5480 reservoirs according to the WCD (2000) minus the very large reservoirs] to total trap efficiency at the European scale. However, on the global scale, Vörösmarty *et al*. (2003) estimated the trap efficiency of very large and large reservoirs at 16 and 12 %, respectively. Using the same ratio between very large and large reservoirs, the impact of all large and very large reservoirs in Europe would be of the order of 40 %. The calculations by Vörösmarty *et al*. (2003) were only based on the hydrological impact that reservoirs have by increasing the residence time, but do not take into account spatial variability in sediment yield. Since the highest sediment yields are found in those European countries with the highest number of reservoirs (Spain, Italy, Romania), they underestimate the true sediment trap efficiency, which is confirmed by the analysis in this study. Furthermore, Vörösmarty *et al*. (2003) do not take into account the impact of the smaller reservoirs. It is clear from the analysis made in this study that reservoirs and ponds have a major impact on sediment delivery at the European scale, probably the most important anthropogenic impact on sediment dynamics that there is.

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2.21

Off-site Impacts of Erosion: Eutrophication as an Example

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2.21.1 INTRODUCTION

Soil erosion is a major global problem, environmentally, economically and socially. Accelerated rates of erosion from agricultural land are rapidly depleting the productive capacity of soils, resulting in losses of agricultural production. In Europe (excluding the Russian Federation), more than 17 % of the total land area is affected by erosion (Gobin *et al*., 2003). In addition to these *on-site* effects, the soil that is detached by water or wind may be transported over considerable distances. This gives rise to *off-site* impacts.

Off-site impacts may be caused by movement of soil itself, examples being sedimentation in water bodies and siltation of lake and reservoir bottoms. Another major off-site impact results from the agricultural chemicals that often move attached to eroded sediment. These chemicals move into, and pollute, downstream watercourses and water bodies. The nature of these off-site problems varies depending on the chemical characteristics; ecotoxicological impacts can often be connected with pesticides, and phosphorus (P) losses may cause eutrophication problems. Moreover, the magnitude of the problem may vary spatially and temporally, depending on local environments. In this chapter we use P loss from agricultural lands as an example of off-site impacts of erosion.

Based on extensive correlation studies between chlorophyll and major nutrients (e.g. Vollenweider, 1968) and on lake experiments (e.g. Schindler, 1977), P has been considered to be the limiting factor for algal growth in freshwater lakes. Thus, the P concentration in many lakes controls primary production and elevated P

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concentrations have often resulted in eutrophication and in various problems connected with it, e.g. potentially toxic algal blooms, proliferation of attached filamentous algae and oxygen deficits. Understanding this cause–consequence relationship launched programmes of intensive control and treatment of waste waters, particularly by municipalities and some industries utilising high P concentration processes. In many parts of the world these abatement measures have been successful and lake water quality has consequently been improved (e.g. Edmondson and Lehman, 1981; Anneville and Pelletier, 2000; Tilzer *et al*., 1991; Häse *et al*., 1998). Owing to the reduction of the point sources of P, non-point sources, particularly from agricultural land currently comprise a significant source of P to waters in many developed countries (e.g. Kronvang *et al*., 1995; Rekolainen *et al*., 1995; Carpenter *et al*., 1998; Van der Molen *et al*., 1998).

The transfer of P from land to waters is controlled by many process (e.g. erosion, leaching) and pathways (e.g. overland flow, subsurface flow, drainflow). In addition, P may enter waters in many forms, inorganic, organic, dissolved, colloidal and particulate. The terminology used to describe P transfer processes, pathways and forms is often confusing; a new terminology was proposed by Haygarth and Sharpley (2000). All P transfer processes, pathways and forms are not directly connected to erosion by water, and all forms of P are not equally harmful in surface waters. Our aim in this chapter is to give an overview of P sources, pathways, forms, impacts and fate in surface waters, and to show that erosion control is important in order to mitigate harmful impacts of P carried by erosion from land to surface waters.

2.21.2 GLOBAL PHOSPHORUS CYCLE

Natural mobilization of P, as part of the grand tectonic denudation–uplift cycle, is slow. Thus on a civilizational time-scale (10^3 years), the natural global P cycle appears to be a one-way flow: mineralisation, weathering, erosion and runoff transfer of P to ocean sediments (Smil, 2000). The global weathering rate has been estimated to be about15 Tg yr^{-1} (see excellent reviews by Smil, 2000, and Bennett *et al*., 2001), and mining for fertilisation (another major input to soils) is estimated to be 18.5 Tg yr^{-1}. Present-day fluvial P flux, the amount of P discharged through rivers to oceans is estimated to be between 18.7 and 31.4 Tg yr^{-1} (Howarth *et al*., 1995; Compton *et al*., 2000), of which only a small fraction (about 2 Tg yr^{-1}) is estimated to be in dissolved form (Martin and Meybeck, 1979; Meybeck, 1982) and between 3.4 and 10.1 Tg yr^{-1} potentially biogeochemically reactive (Compton *et al*., 2000) Thus, erosion is the major process carrying P from land to oceans.

Conversion of forests and grassland to croplands has increased erosion. Smil (2000) estimated that the present P erosion rates are about twice as high as pre-industrial erosion. Moreover, large-scale use of inorganic fertilisers has resulted in the accumulation of P in agricultural soils, which is currently ca 11.5 Tg yr^{-1}, and more than doubled during the past 40 years (Bennett *et al*., 2001). Often the excess fertiliser P concentrates in the surface soil layers, enriching erodable soil by P.

2.21.3 TRANSPORT MECHANISMS AND PATHWAYS

2.21.3.1 Flow Pathways

Phosphorus transport starts through the action of chemical, biological and biochemical processes that together define the mechanisms of ‘solubilisation’ or ‘detachment’ depending on the physical size of the mobilised P compounds (Haygarth *et al*., 2005). Solubilisation is the release of molecules or macromolecules of P from soil surfaces and soil biota into soil water. There is evidence that the potential for P solubilisation increases with increased soil P status in runoff (e.g. Heckrath *et al*., 1995); see Section 2.21.4.3. Detachment of

soil particles and attached P is a physical mechanism for mobilising P from soil into waters. Phosphorus transport from the point of mobilisation at the soil profile scale to the point of input to receiving waters at the landscape scale is complex because it embraces a range of spatial and temporal scales that expand from the point of mobilisation, and must include ensuing changes in P forms. Further complexity arises because P can be deposited and entrained during transport. The main hydrological pathways of P transport are described below.

2.21.3.1.1 Surface Pathways

Two starting points for the generation of surface runoff are recognised. The first, infiltration-excess flow, is generated when the infiltration capacity of the surface soil is exceeded, usually as a result of high-intensity storm events. The second, saturation-excess surface runoff, is topographically driven from spatially and temporally dynamic variable source areas. This pathway is triggered where the soil becomes saturated via lateral percolation above an impeding horizon. Saturation-excess surface runoff also occurs where the soil water table rises to the ground surface through convergent flow into hillslope hollows or a rising stream water level results in saturation of near-stream zones. Under steady rainfall, saturation-excess flow requires much lower rainfall intensities to maintain it in comparison with infiltration-excess flow and is generally a more important mechanism of surface runoff generation.

2.21.3.1.2 Subsurface Pathways

Subsurface flow may reach the drainage network via a number of pathways: (i) groundwater, (ii) lateral flow where soil layers have vertical conductivity $<$ rainfall intensity and (iii) where concave topographic contours create contributing areas because a high water table and/or subsurface impedance causes convergent flow. Where soils are deep and the bedrock permeable, percolation to groundwater rather than channelling of flow laterally will occur. The rate of subsurface flow depends on soil hydraulic conductivity, which defines whether matrix flow (saturated/piston flow) or preferential (macropore) flow predominates. Preferential flow defines a rapid pathway of water transit through the soil. Some soils, such as cracking clays, have a greater preponderance of macropores and hence more channelling of subsurface flow via this pathway.

2.21.3.2 Phosphorus Forms in Flow Pathways

In general, and primarily for tilled land, P transport in particulate form is associated with surface runoff. Here the selective adsorption of P on to clay and silt-sized soil particles (as Fe/Al complexes or Ca/Mg phosphate) allows mobilisation with soil eroded from agricultural land. Transport of P in particulate organic form is important is grassland systems (Heathwaite *et al*., 1990). Subsurface pathways are commonly associated with P transport in dissolved form. However, preferential flow may also be an important pathway of particulate P transport (Dils and Heathwaite, 1996; Heathwaite, 1997), particularly attached to colloidal material (Heathwaite *et al*., 2005).

2.21.3.2.1 Surface Pathways of P Flow

Surface runoff has a strong affinity for P transport because the surface soil has the greatest effective depth of interaction (EDI) with rain and snowmelt water and the highest concentrations of P. Phosphorus residing in the surface 0.5 mm of soil appears to be most vulnerable to export in runoff. Phosphorus transport in surface runoff is influenced by farming type, erosion potential, hydrologically effective rainfall, land use including fertiliser and manure amendments, the presence or absence of livestock and soil total P (Heathwaite, 1997).

Surface runoff is important in physically transporting P via soil erosion. Even where erosion is minimal, elevated soil P can sustain high TP losses. For example, for grassland soils P transport in surface runoff may be exacerbated by high P concentrations at the soil surface as a result of organic matter inputs (Haygarth *et al*., 1996).

Where land management has increased, the incidence of infiltration-excess surface runoff, significant transport of P may occur during storm events – often on a field-wide scale (Heathwaite, 1997). Some land management practices or crop types present a greater risk of P transport than others. Where P transport is linked to soil erosion, high-risk crops include winter cereals and winter vegetables, with temporary grass (<5 years old), potatoes, sugar beet and maize of medium risk and other arable crops such as spring cereals and oilseed crops of low risk (Chambers, 1997). In the UK, around 60 % of high-risk land is actually eroding, with 20 % for medium-risk land and 10 % for low-risk land (Chambers *et al*., 1992). The authors estimate that only 33–43 % of erosion events actually transport sediment and P to streams. Thus, on average, for the UK, around $5000\,\mathrm{t\,yr^{-1}}$ or $0.6\,\mathrm{kg\,total\,P\,ha^{-1}\,yr^{-1}}$ may reach watercourses. Land management such as grazing fodder crops may compact the soil surface and decrease the infiltration capacity of the surface soil, leading to sheet erosion and associated P transport on a field-wide scale (Heathwaite *et al*., 1990). Chambers (1997) suggests that P loss by sheet erosion could be significant in the UK because, when triggered, it operates over large land areas.

2.21.3.2.2 Subsurface Pathways of P Flow

Three subsurface pathways are recognised as having potential for P transport. First, near-surface lateral flow, owing to higher soil P concentrations in upper soil horizons – although P present in this horizon may not necessarily be mobile and appears to be dependent on the mechanism of subsurface flow. In general, matrix flow is unlikely to initiate significant P transport, whereas preferential flow may be important. The significance of P transport to groundwater by leaching depends on the depth to the water table and P application at the soil surface. In The Netherlands, for example, the shallow water table and high P loading at the soil surface have created a high potential for P transport to groundwater. Here, van Riemsdijk *et al*. (1987) suggest that breakthrough of high P concentrations to groundwater is likely within 20–30 years if manure P loadings at the soil surface continue at current rates. Second, preferential flow may enable rapid subsurface transport of mobile P through soil macropores. Macropore flow reduces the time for interaction and hence the degree of transformation of P forms during transit. This may affect the bioavailability of P reaching the stream network. In addition to dissolved P forms, this pathway may be important for P transport in particulate and colloidal form – particularly from grassland soils (Heathwaite, 1997). Dils and Heathwaite (1996) found that around 68 % total P transport in macropore flow from a mixed grass/arable catchment was in the particulate fraction where the organic phase dominated, accounting for around 62 % total P transported in macropore flow in the upper 45 cm of soil. Third, artificial drainage acts like preferential flow to encourage rapid transit of water from land to stream. Phosphorus loss in drainflow is influenced by soil type (stability), soil total P and excess winter rainfall (Chambers, 1997). Drained clay soils, for example, transmit water rapidly via cracks and mole channels; contact with subsoil is minimal and high P losses might be anticipated, particularly where such soils receive high fertiliser or manure amendments. Kronvang *et al*. (1997) found that up to 18 % of annual particulate P loss from a lowland arable catchment in Denmark was transported in subsurface drainage. Total P loss from grazed underdrained land with high animal manure inputs was over five times greater than underdrained arable catchments (0.63 and $0.12\,\mathrm{kg\,P\,ha^{-1}\,yr^{-1}}$, respectively) (Grant *et al*., 1996).

P transport does not stop with entry to receiving waters but continues instream. Here, the extent, nature and dynamics of interactions between soluble and particulate P in water and sediments play a critical role in the impact of P on the aquatic ecosystem.

2.21.4 AVAILABILITY OF P

2.21.4.1 Estimating Algal Availability of Different P Forms

Despite numerous studies – starting from Golterman *et al*. (1969) – it appears that no consensus has been reached on the question of whether all P is equally important in eutrophication, or if there are differences among the forms of P in terms of their utilization by aquatic biota, planktonic algae in particular (e.g. Sonzogni *et al*., 1982; Correll, 1998; Reynolds and Davies, 2001). The question of P availability has many implications on the assessment and control of eutrophication. These cover the determination of P, dose–effect predictions and P reduction targets and measures.

Planktonic algae take up P mostly in the form of dissolved orthophosphate (Cembella *et al*., 1984). The availability of other P forms thus depends on the rate and extent to which they are transformed into dissolved orthophosphate. Potentially algal-available P has been defined as the sum of dissolved orthophosphate and the P that can be transformed into this directly available form by naturally occurring physical, chemical and biological processes (cf. Boström *et al*., 1988). As such, it is a measure of the upper limit of the algal-available P (Sonzogni *et al*., 1982).

The loss of P from terrestrial systems is a complex phenomenon and occurs in dissolved and particulate forms (see Sections 2.21.3 and 2.21.4.2). In routine monitoring, it is customary to analyse two P fractions from runoff: total P (TP) and dissolved (molybdate) reactive P (DRP), the latter being determined from a filtered sample (Broberg and Persson, 1988). Particulate P, which largely represents P in eroded soil, is then obtained by subtracting DRP (or dissolved total P) from TP. Although DRP is not equivalent to dissolved orthophosphate (Broberg and Persson, 1988), it is generally considered readily available (Lee *et al*., 1980; Ekholm and Krogerus, 2003). The potential availability of particulate P in runoff has been tested by laboratory bioassays for the utilisation of P by (mostly) monocultures of algae (e.g. Dorich *et al*., 1980; DePinto *et al*., 1981). Such assays have sometimes been replaced – with varying success – by relatively simple chemical methods, such as extractions with 0.1 mol l^{-1} NaOH (Cowen and Lee, 1976; Dorich *et al*., 1985; Sharpley *et al*., 1991), iron oxide-impregnated filter-paper (e.g. Ekholm and Yli-Halla, 1992; Sharpley, 1993; Dils and Heathwaite, 1998) and ion exchangers (Cowen and Lee, 1976; Hanna, 1989; Uusitalo and Yli-Halla, 1999).

According to algal assays, the availability of P in eroded/erodible soil varies widely, partly depending on the origin of the matter. For example, only <1–13 % of P in eroding river banks was found to be available by Ellis and Stanford (1988) and P in eroding lake bluffs was reported to be entirely unavailable by Young *et al.* (1985). On the other hand, the mean availability of particulate P in agricultural rivers has ranged from 5 % (Ekholm, 1994) up to 41 % (Persson, 1990), most of the results falling in the range of about 20–30 % (Dorich *et al*., 1980; Williams *et al*., 1980; DePinto *et al*., 1981; Källqvist and Berge, 1990). The corresponding values for forested rivers have ranged from 6 % (Ellis and Stanford, 1988) to 55 % (Persson, 1990). Hence there appears to be some difference in algal availability between natural and human affected P sources. However, it should be noted that generalisation of the results is problematic owing to considerable methodological diversity among the studies (Rekolainen *et al*., 1997).

Ekholm and Krogerus (2003) tested a large variety of samples for potentially algal-available P using the same method (dual culture algal assays). Particulate matter in agricultural runoff appeared to have a higher mean proportion of algal-available P in total P (17 %) than forest runoff (4.5 %), an observation attributable to the markedly higher fertilisation rates applied to agricultural soils. The availability of P in cultivated top soil was similar to that in the particulate fraction of field runoff but higher than in agricultural rivers. The lowest availability was found in lacustrine particles, possibly because most of the available P had already been mobilised from these particles during transport to – and finally in – the P-limited recipients.

Although bioassays have been considered the most reliable technique to estimate algal-available P (e.g. Boström *et al*., 1988), it is probable that they underestimate the true potentially algal-available P. For example,

compared with the residence time of P in aquatic systems, the incubation time is usually short (a few weeks). In addition, P availability may vary between different types of receiving waters as a function of their – seasonally varying – physical and chemical conditions (Dorioz *et al*., 1998; House *et al*., 1998). It should be especially emphasised that the results from algal assays are applicable only to aerobic systems. When eroded soil finally becomes part of the bottom sediments of a water body, it will gradually by burial enter zones with a redox potential sufficiently low to allow the well-documented phenomenon of Fe reduction with concomitant P release. Uusitalo and Turtola (2003) simulated such conditions by adding buffered dithionite to turbid runoff from clayey fields. They found that it induced 2–3 times higher P release than found in oxic conditions. Whether the P thus liberated will enter the productive water layer or be recaptured by surface sediment will determine its importance in eutrophication. Since marine sediments appear to have a lower P-binding capacity than freshwater sediments (e.g. Jensen *et al*., 1995), the importance of Fe-bound P may be accentuated in marine systems (see Section 2.21.5). Finally, it is also possible that the estimates obtained by algal assays on P availability exceed the actual availability. Such may be the case in deep oligotrophic lakes in which particulate matter is settled relatively rapidly – and permanently – out of the productive water layer (Berge and Källqvist, 1998).

As was shown by Hens and Merckx (2001), application of many different techniques for the same samples is a very powerful approach in studies aiming at revealing which compounds are involved in, and affect, the reactions of P in different circumstances. These types of studies should combine several sample modification techniques such as isotope exchange, spikes with anions competing for P sorption sites, specific dissolution of some of the compounds taking part in P exchange [e.g. redox treatments for Fe(III) oxide dissolution], destruction of part of the P associations (e.g. organic matter by UV light), enzyme studies, bioassays, etc, with the ability of, e.g., gel filtration chromatography, or the more simple DGT (diffusive gradients over thin films) technique, to separate ortho-P from other small-sized molybdate-reactive P associations. Of course, these techniques are only applicable for a limited number of samples. For large surveys, other methods developed for the assessment of potentially bioavailable particulate P should be applied, as these data are few (one such study is shortly summarized in Section 2.21.4.3).

As a conclusion, it appears that the availability of eroded P is dependent, on the one hand, on the origin of the matter and, on the other, on the recipient it enters. Methodology for the estimation of potential algal availability of P is, however, still defective, which confounds the reliable estimation of the role of eroded P in eutrophication.

2.21.4.2 Transformation of Eroded P into Dissolved Orthophosphate

Even though particulate P is less bioavailable than DRP (see above), characterisation of the tendency of particulate P (PP) to transform into a bioavailable form is emphasized by its high share of total P losses in a variety of cropping systems (e.g. Culley *et al*., 1983; Djodjic *et al*., 2000; Ng Kee Kwong *et al*., 2002). The highest relative shares of PP are likely found in annually cultivated sloping soils planted with row crops, but also grassed fields or pastures may deliver significant portion as PP (Fleming and Cox, 2001; Hooda *et al*., 1999; Turner and Haygarth, 2000). Of the P transport by the world rivers, about 90 % occurs as attached to suspended particles (Martin and Meybeck, 1979), and the flux of PP that may desorb in the oceans has been estimated to be 2–5 times the riverine DRP flux (Froelich, 1988).

Solid-phase P transformation into dissolved ortho-P is brought about by interacting chemical and biological processes. In describing binding and release of P in soils and sediments, the following components are included: (i) P sorption on to and desorption from Al and Fe oxides (Harter, 1968; Hsu, 1964; Richardson, 1985), (ii) precipitation, coprecipitation or sorption on to $CaCO_3$ and dissolution of secondary Ca–P associations (House and Donaldson, 1986; House *et al*., 1998; Mattingly, 1975), (iii) dissolution of Fe–P

associations and formation of Fe–P precipitates under varying redox environments (Li *et al*., 1972; Miller *et al*., 2001; Gunnars *et al*., 2002), (iv) weathering of primary P-bearing minerals (e.g. apatite) (Walker and Syers, 1976) and (v) formation and mineralisation of organic P-bearing compounds (Walker and Syers, 1976; Benitez-Nelson and Buesseler, 1999; Paytan *et al*., 2003).

The most relevant PP fractions in plant nutrition and eutrophication points of view are those associated with Al and Fe oxides (Froelich, 1988; Muljadi *et al*., 1966a; Richardson, 1985). Phosphorus associated with primary P-bearing minerals can be taken as an extremely long-term P reserve, and often considered as refractory P sources for algae (Williams *et al*., 1980). Even though organic P can be utilised by algae or bacteria (e.g., Benitez-Nelson and Buesseler, 1999; Suzumura and Kamatani, 1995), its role in eutrophying P load is less well documented as compared with inorganic non-apatite P. In a eutrophication context, the above may be summarised by stating that the potentially algal-available portion of PP mainly consists of inorganic non-apatite P (Logan *et al*., 1979; Williams *et al*., 1980).

In-field processes of PP transformation into DPR are mainly limited to desorption and dissolution. During rainfall events or irrigation, these reactions are enhanced by an increase in water with low ortho-P concentration. In the case of dissolution, decreased Ca and ortho-P concentrations in the water phase reverses the formation of secondary Ca phosphates into solubilisation of these compounds.

For desorbable PP, an increased water-to-solid ratio disturbs the (semi-)equilibrium between solid and solution-phase P, and a new equilibrium which corresponds to lower solution P concentration is approached (Barrow, 1979; Hartikainen, 1991; Delgado and Torrent, 2001; Torrent and Delgado, 2001). Desorption of PP will continue as long as there are changes in solid-solution P equilibria, and if the eroded particles come into contact with higher solution P concentration than they have equilibrated to, sorption of P on to particle surfaces may also occur (Sharpley *et al*., 1981). As P desorption from oxide surfaces is a ligand-exchange reaction (Hingston *et al*., 1967) in which ortho-P and OH^- participate, changing pH environment also affects P sorption/desorption reactions in soils and sediments (e.g. Muljadi *et al*., 1966b; Koski-Vähälä and Hartikainen, 2001). Substantial changes in pH environment, and also redox environment, however, more often occur in receiving waters and are described in Section 2.21.5.

2.21.4.3 Dissolved and Particulate P in Runoff

Desorption of P into water and hence typical DRP concentration in edge-of-field runoff are governed by P surface saturation of the soil layer interacting with runoff water (Hooda *et al*., 2000). Most of the runoff DRP originates from the soil that remains in the field (Yli-Halla *et al*., 1995), even though desorption of runoff sediment PP starts at the same instance as the soil particle is detached from field surface. Depending on soil P status and the properties of the receiving waters, runoff sediment may then have significant potential to supply P in bioavailable form. The amount of Al and Fe oxide-associated (desorbable and redox-labile) runoff sediment P seems to be directly related to the reserves of plant-available soil P, such as measured by using the Olsen or Mehlich P tests (Uusitalo *et al*., 2003).

Regarding soil loss from agricultural fields, the P content of eroded soil matter can be assumed to be greater than in pristine soil and streambank sediment, and soil matter eroded from agricultural soils is therefore rather a source than a sink for bioavailable P. Eroded sediment may also contain more P than its source (Sharpley, 1985), especially if soil texture is not uniformly fine, because with decreasing particle size the P content of mineral particles increases (Syers *et al*., 1969; He *et al*., 1995; Maguire *et al*., 1998; Øgaard, 1996), in addition to the ease of particle transport with runoff. An example of geographical variability in suspended sediment P content, brought about by variations in parent rock P content and pedogenesis, can be found in Martin and Meybeck (1979), who reported a range between 540 and 2000 mg P kg^{-1} for suspended sediment collected from nine large rivers in different parts of the world. However, also in geologically uniform areas agricultural

P management may produce similar variations in the P contents of suspended runoff sediment (Uusitalo and Jansson, 2002) or in the P contents of sediment depositions in waterways (Verstraeten and Poesen, 2002), which is then also affected by sorting of eroded matter by size and density during deposition.

Applying anion-exchange resin extraction (for desorbable PP) and redox treatment (solubilisation of redox-labile P associations, such as Fe hydroxides) for edge-of-field runoff samples, Uusitalo *et al*. (2003) estimated that eroded soil matter from four Finnish clayey soils would deliver as much algal-available PP as was lost as DRP. In turn, the annual losses of redox-labile PP were 2.4–3.0 times the DRP losses. The estimated algal availability of PP at the four field sites studied was 10–20 %, or 95–305 mg P kg^{-1} runoff sediment (recalculated to account for underestimation of algal availability of PP by anion-exchange resin extraction; Uusitalo and Ekholm, 2003), whereas redox-labile PP was assessed to account for 34–58 % of runoff PP, which would correspond to 330–830 mg P kg^{-1} sediment matter. Compared with maximum bioavailability estimates of streamwater PP, the above estimates were lower than the 51–73 % bioavailability reported by Pacini and Gächter (1999) and 60–75 % by Dorich *et al*. (1985) and James *et al*. (2002). The latter assessments were based on NaOH-extractable PP, as such or as part of sequential extraction schemes. Logan *et al*. (1979) reported that NaOH-extractable PP, as an estimate of short-term bioavailability, would account for 14–42 % of sediment-P (total P: 720–1200 mg kg^{-1} sediment) collected from US streams draining into Lake Erie. In turn, non-apatite inorganic P accounted for 43–89 % of sediment-associated P.

In artificially drained soils, subsurface runoff is an important pathway for water and solutes, and attention has increasingly focused on subsurface drainflow as a contributor to the P load from agricultural fields and pastures (Hooda *et al*., 1999; Laubel *et al*., 1999; Øygarden *et al*., 1997). With regard to PP, Turtola and Paajanen (1995) and Simard *et al*. (2000) reported that drainflow was relatively enriched with PP, compared with surface runoff. It has been shown by Cs-137 analysis that much of the drainflow sediment may derive from the topsoil (Laubel *et al*., 1999; Uusitalo *et al*., 2001). In addition, the content of anion-exchange resin-extractable PP in drainflow sediment in clayey soils has been found not to differ from that measured in sediment delivered by surface runoff (Uusitalo *et al*., 2001). When much of the water runs off tile-drained fields via subsurface pathways, the above results indicate very high potential for the transport of bioavailable PP via subsurface flow, in some sites far greater than via surface runoff (Uusitalo *et al*., 2003). In these situations, effective erosion control would require means to reduce detachment and dispersion of soil at the entire field surface, not only changing the flow pathways and establishing buffer strips in field margins.

2.21.5 FATE OF P IN FRESHWATER AND BRACKISH–MARINE WATERS

After entering an aquatic system, from any source or in any of its many possible forms, P will take part in a complex biogeochemical cycle. However, ultimately most of the P will be buried in the bottom sediments. After settling, particulate P begins to undergo chemical and biological reactions, which lead to its solubilisation. Two mechanisms have to occur simultaneously or within a short period of time before P is released from sediment to water: first, the dissolution of P bound to particles and transfer to the pool of dissolved P in the pore water, and second, a transport process responsible for the release of dissolved P from sediment to water (Boström *et al*., 1982).

Ferric iron [Fe(III)] – existing mainly as various oxides and oxyhydroxides – has a high ability to bind P. When Fe(III) oxide is reduced to ferrous iron [Fe(II)], it loses the P-binding ability. In sediments, the reduction of Fe(III) oxides occurs by two main mechanisms (Lovley, 1991): first, in microbial dissimilatory reduction of Fe(III) oxides; and second, in chemical reduction by hydrogen sulfide (H_2S).

In sediments, the classical P cycling model is based on Fe cycle in the sediment: insoluble Fe(III) oxides are reduced to soluble Fe(II) ions and after the reduction P bound to Fe(III) oxides is released into pore water. When reduced, soluble Fe(II) is diffused to an oxic layer a few millimetres thick in the sediment surface

(Jørgensen and Revsbech 1985), and it is chemically and microbially oxidised to Fe(III) oxyhydroxides. The precipitated Fe(III) oxyhydroxides present in the sediment surface layer prevent effectively P from entering the water column (Tessenow, 1975; Mayer *et al*., 1982; Gallagher, 1985; Ellis-Evans and Lemon, 1989).

Based on laboratory experiments and measurements from aquatic systems, a molar ratio of 2 between dissolved Fe(II) and DRP is suggested to be sufficient to retain P in sediments, when conditions turn from anoxic to oxic (Gunnars and Blomqvist, 1997; Gunnars *et al*., 2002). Below a ratio of 2 there is not enough Fe(II) to form Fe(III) oxyhydroxides to bind all diffusing DRP in the oxic zone.

In the sediments the cycling of Fe, however, may be inactivated by S. From S compounds it is the SO_4^{2-} anion that probably dictates the major difference in the reduction and burial of Fe between the freshwater and the brackish–marine sediments. For example, the average SO_4^{2-} concentration of 2.8 mg l^{-1} in Finnish lake waters (Mannio *et al*., 2000) is almost 180-fold lower than that of the brackish Gulf of Finland, Baltic Sea ($\sim$500 mg l^{-1}; Lehtoranta, 2003).

It is likely that in organic-rich brackish–marine sediments Fe(III) oxyhydroxides are reduced mostly by H_2S, because SO_4^{2-} reduction is the dominant anaerobic respiration process. Ultimately, even very stable crystalline Fe(III) oxides are converted to pyrite (FeS_2) in sulfidic sediments (Canfield, 1989). For example, in Århus Bay, Southern Baltic, partial oxidation of H_2S has accounted for 63 % of the estimated Fe(III) reduction, whereas contributions of microbial Fe reduction to C mineralisation were considered to be small (Thamdrup *et al*., 1994). Therefore, in brackish–marine sediments the anaerobic microbial SO_4^{2-} reduction forms large amounts H_2S, which is capable of effectively reducing Fe(III) oxyhydroxides. As a consequence of the reduction by H_2S, the Fe(III) oxide-bound P is released into the pore water, whereas Fe is bound as solid FeS, which binds P insignificantly (Bebie *et al*., 1998). Consequently, the dissolved Fe:DRP ratio is almost invariably below 2 in anoxic conditions in brackish–marine systems (Gunnars and Blomqvist, 1997).

The efficient FeS formation and burial prevents the upward diffusion of Fe and, thus, the formation of Fe(III) oxyhydroxide layer in the oxic zone of the sediment. As a consequence, DRP is almost freely transported into the water column and thus the insufficient ability of formed Fe(III) oxyhydroxides to bind DRP partly explains the high efflux of DRP to oxic water (Gunnars and Blomqvist, 1997; Lehtoranta, 2003). Additionally, when the molar Fe:P ratio is low (i.e. 0.03–0.2; Lehtoranta, 2003), it is evident that other sediment mechanisms (e.g. rate of formed DRP and microbial uptake) rather than the sorption of P on to formed Fe(III) oxyhydroxides largely control the P release from sediment to water. The ability of brackish–marine sediments to retain P, however, may be high when the dissolved Fe:DRP ratio exceeds 2 in surface pore water (Lehtoranta, 2003).

In general, the scavenging capacity of P is better in freshwater systems than in brackish–marine systems. This can be explained by the fact that there are high amounts of dissolved Fe in relation to P in the freshwater systems, whereas in the brackish–marine systems the amount of Fe is not sufficient to bind all the DRP in solution when anaerobic conditions turn to aerobic.

The principal N limitation in brackish and coastal marine systems may be attributed to the low P binding ability of their bottom sediments (Caraco *et al*., 1989; Gunnars and Blomqvist, 1997; Roden and Edmonds 1997). In freshwater sediments, Fe provides an efficient sink for P, resulting in a shortage of this nutrient. The Fe(III) oxide-bound P may be less bioavailable in freshwater than in brackish–marine systems and, therefore, the erosion control of particulate P may be more valid in brackish–marine systems.

2.21.6 IS EROSION CONTROL WHAT WE NEED TO CONTROL P LOSS AND EUTROPHICATION?

In addition to P, eutrophication is controlled by many other factors (e.g. light attenuation) and nutrients (particularly nitrogen, which most often is the growth-limiting factor in marine waters). However, P has a key role in accelerated eutrophication and its harmful socio-economic consequences worldwide. Analogously,

erosion control is not the only cost-effective mitigation action in order to reduce P input to waters. Others are needed, in many regions and countries the P flux through municipal waste waters still comprises the major source of P, and in developing countries it may still be increasing with increasing human population. Nevertheless, P flow from bedrock (either by weathering or mining) through agricultural soils to waters is a major transport route of P, resulting in undesired impacts both in fresh and marine waters.

Many studies, at global, regional and local scales show that most of the P from agricultural soils enters waters in particulate form, thus being detached and transported by erosion processes. This review indicates clearly that a remarkable fraction of this particulate P might be utilisable by algae, and, on the other hand, much of the dissolved P fraction might originate from erosion processes.

Erosion mechanisms and transport pathways in soils are, however, complicated, interactive and often nonlinear. Successful design of cost-effective control measures requires knowledge of these mechanisms, and pathways and also of the structure of the production and management systems. Often, the best results can be obtained by a mixture of various measures, some affecting the source (e.g. balanced fertilisation, farm management systems) and others being end-of-pipe measures (e.g. buffer strips, re-establishment of wetlands and flood plains).

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2.22

Economic Frame for Soil Conservation Policies

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2.22.1 INTRODUCTION

Reflecting the economic effects of soil erosion, a wide range of different aspects have to be considered. First, it might be helpful to narrow the issue down to what Evans (1993) called 'accelerated erosion', i.e. the human contributions to soil degradation. Accelerated erosion is primarily related to agricultural activities. Therefore, from an economic viewpoint, soil erosion can be understood as the negative external effects or as the external costs of agricultural production. These costs may stem from different impacts. First, erosion leads to degradation of farm land. From a farmer's point of view, this might be reflected in a loss of productivity and, as a consequence, a decline of income or of the farm's resale value (McConnell, 1983). However, it might also cause costs from a societal point of view, when society attributes, e.g., existence values to the mere availability of well-maintained soils. Second, soil erosion leads to off-site impacts, where the problems of water pollution and other effects such as muddy floods and blocked dams are paramount (McConnell, 1983; Pretty *et al*., 2001). Hence the assessment boundaries of the economic effects of soil erosion have to be expanded at least to the issue of water pollution. Cost–benefit analysis is an approved methodological approach for carrying out economic assessments within the framework drafted above.

According to Pretty *et al*. (2001), cost–benefit analysis may be carried out on at least two different aggregation levels. One focuses on the national or international level where policy strategies are developed. Investigations concentrate on estimating the total costs, both overall and by type of externalities, e.g. the cost of eliminating water pollution caused by soil erosion. The second level focuses on the economic assessment of specific policies or conservation programmes. The estimations of private and social costs and benefits can help to guide decisions on policy or programme development, e.g. which programme specification is best suited to

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achieve a given level of soil erosion in a water catchment area with minimal costs. This chapter will concentrate on the first level of national and international policies. Regarding the second level of specific programmes, the reader may refer to Hanley *et al*. (1999) and Oglethorpe *et al*. (2000).

Cost–benefit analysis provides important information for decision-makers about the economic rationality of a political strategy. It delivers scarce information about options to react to soil degradation processes or to steer political processes towards an implementation of soil erosion strategies. To attain an understanding about political processes, institutional economics and political economics provide consolidated findings. Most papers published in connection with economic effects of soil erosion do not reflect adequately on these important issues (e.g. McConnell, 1983; Pretty *et al*., 2000, 2001; Evans, 2002; Boardman *et al*., 2003). However, basic knowledge of the main principles of political economics may help stakeholders engaged in soil erosion to gain an understanding of why the implementation of soil conservation policies lags far behind the scientific perceptions, and which crucial factors must be taken into account to implement soil conservation strategies successfully. Therefore, the second part of this chapter discusses the topic from the viewpoint of institutional and political economics. Here, we will concentrate on the theoretical framework of property rights and transaction costs.

Finally, in addition to economic theories, a number of socio-economic variables play an important role with respect to the implementation of soil conservation strategies. According to Ervin and Ervin (1982), primarily personal attitudes (family, ethnicity, religion) towards soil conservation play an important role. Additionally, sociological concepts of community information processes might be relevant regarding the uptake of conservation practice (e.g. effectiveness, early/late adopters) (Ervin and Ervin, 1982, p. 290; see also Rogers, 1983; Albrecht *et al*., 1987). As this chapter focuses on economic considerations, interested readers may refer to the corresponding sociological literature.

2.22.2 ECONOMIC ASSESSMENT OF SOIL EROSION

Soil erosion affects economic concerns on both a private and a societal level. Even though emphasis is on off-site impacts, a short discussion of private on-farm effects is presented out to complete economic considerations of soil erosion.

2.22.2.1 On-farm Effects

On-farm considerations deal mainly with long-term productivity effects of erosive land and the anticipated effect on land productivity (Ervin and Ervin, 1982; Evans, 1993). Because of methodological difficulties with such long-term considerations, there are, at least for the example of Western Europe, only a few studies that provide empirical data on long-term economic on-farm effects of soil erosion (Evans, 1995). However, several authors have analysed the problem from a theoretical point of view (McConnell, 1983; Dabbert, 1994). One of the major outcomes of these analyses is that given certain circumstances, it is economically rational to accept a certain amount of soil degradation. Theoretically, assuming perfectly functioning market mechanisms, a farmer's economic rationale would lead to the integration of soil conservation measures into private decision making by matching marginal costs of soil erosion with the marginal costs of soil conservation. According to McConnell (1983), farmers are self-interested in soil conservation because they want to maximise the total income of their farms. This total farm income consists of the value of the annual profits, achieved by agricultural activities but also by the farm resale values. As farmers know very well that the soil base affects farm resale values, they will arrange for conservation measures. McConnell's approach was strongly questioned by Kiker and Lynne (1986), who showed that market signals do not guarantee proper conservation of soils according to societal needs.

TABLE 2.22.1 On-farm costs of erosion in England and Wales (based on 1991 prices)

		£million per year
Lowlands		
Water erosion	Lost inputs	0.499
	Lost outputs	1.644
Wind erosion	Lost inputs	0.368
	Lost outputs	1.232
Uplands		
	Lost outputs	1.079
	Value lost riparian land in Wales	3.468–4.224
	Total	8.290–9.046

Source: Evans (1995).

Empirically, McConnell's approach also could not be proved, since the assumptions on perfect markets (i.e. full information about the impact of past and future damage) can easily be questioned. The resale value of agricultural land was not influenced by the erosion-related condition of topsoils (Ervin and Mill, 1985), which leads to the assumption that soil conservation is of less concern to an individual than to society.

Within these considerations, the level of interest rates plays a major role. When interest rates are high, the present value of the farm resale value in comparison with the annual profits is low and soil conservation does not play a major role in farmers' decisions and *vice versa*. As a result, farmers' readiness to implement soil conservation measures depends also on land prices, which are strongly connected with prices of agricultural products (Dabbert, 1994).

Empirical studies of on-farm costs of soil erosion were conducted in England and Wales by Evans (1995). According to this study, short-term aggregated on-farm costs of soil erosion amount to about £4.8 million (€7.1 million) plus approximately £3.9 million (€5.77 million) for lost riparian land in Wales, both at 1991 prices (see Table 2.22.1).

In conclusion, theoretical and empirical studies suggest that economic on-farm effects of soil erosion do exist, but are still not high enough to force farmers to adopt soil conservation strategies to an extent that might be optimal from a societal or environmental viewpoint. Uncertainty about the long-term damage of soil erosion and the benefits of conservation measures have an influence on farmers to realise such a strategy.

2.22.2.2 Off-farm Effects

Society may allot to soils, as a part of landscapes, other values than the mere use value (as a production input). These include environmental and scenic values, the provision of space for infrastructure (roads, railways, etc.) and resources (drinking water), and the long-term perspective of soil use as a resource for food security.

In these cases, society has to bear the costs of soil degradation since its goods and services are affected. Usually, the economic literature discusses off-farm effects under particular considerations of external effects with a focus on off-site damage such as flooding or pollution of drinking water with soil components such as sediment, organic carbon and nutrients (Pretty *et al*., 2000, 2001; Evans, 2002; Boardman *et al*., 2003). In these studies, the terms 'off-farm' and 'off-site' are mostly used synonymously. However, economic assessment of off-farm effects goes beyond the valuation of off-site effects, i.e. off-farm and off-site effects are not synonymous. Off-farm effects occur outside the farm understood as an economic unit, whereas off-site effects occur outside the field understood as a spatial unit. Below, a broader approach is presented that considers off-farm effects which may occur both on-site and off-site.

TABLE 2.22.2 Components of the 'Total Economic Value' of soil resources

Total Economic Value					
Use value				Non-use value	
Direct value	Indirect value	Option value	Primary value	Existence value	Bequest value
Agricultural and silvicultural production Science Education Etc.	Flood protection Pollutant sink Monitoring	Direct and indirect values for future generations	Functions for the ecosystems	Intrinsic value Cultural value	Heritage for future generations

Source: modified from Pearce (1993).

2.22.2.2.1 The Total Economic Value of Soil and Soil Conservation

According to Pearce (1993), the 'total economic value' (TEV) of a nature resource is the sum of use values and non-use values. The use value consists of direct and indirect values. The direct value expresses, e.g., the production function, the indirect value that expresses (i) soil ecosystem functions associated with, e.g., flood protection, cleaning, buffering, storage, cooling, habitat functions, and (ii) the option value. The latter corresponds to the direct and indirect value for future generations, which is the value to maintain options for future generations, e.g., to grow food and fibre in a satisfactory way. The non-use value describes the existence value of soils, i.e. society's appreciation of the mere existence of different soil types (intrinsic value) that is beyond conventional utilitarian approaches. Turner and Jones expand the TEV approach by introducing the primary value that expresses the ecosystem functions of soil (cited in Pearce, 1993). According to Tisdell (1991), humans desire to leave sound soil resources for future generations that includes, *inter alia*, the cultural heritage function of soil that can also be evaluated and expressed as a bequest value. A brief aggregation of these considerations is given in Table 2.22.2.

These theoretical considerations show that the value of soil goes far beyond its value for agricultural or silvicultural production. Even though theoretical evidence is high, it is very difficult to quantify the societal or aggregated individual benefit for most of the soil value components.

As a substitute for assessing single value components, economists have developed indirect methods to estimate the value of soil conservation. One starting point to obtain an idea of what soil conservation may be worth is to assess the costs that society has to bear in case of soil erosion. As already mentioned, there are some advanced investigations concerning this approach based on the estimation of off-site damage caused by accelerated soil erosion.

2.22.2.2.2 Investigations to Assess the Value of Soil Conservation by Assessing Costs of Off-site Damages

One of the most common methods to evaluate economic effects of soil erosion is to measure negative external effects that occur in connection with it. According to Baumol and Oates (1988), external effects occur if the benefit of an individual or institution A is affected by activities of an individual or institution B, that does not include these welfare effects consciously within his decision making. Second, external effects are characterized by the lack of a mechanism to compensate for positive or negative effects between individuals B and A. Pretty *et al*. (2001) describe disrupting water courses and runoff from eroded land causing flooding and damage to housing and natural resources as examples of negative external effects of soil erosion.

There are few investigations that quantify negative external effects in case of soil erosion and none that refers to the entire area of Europe. Before presenting some examples, it is important to clarify the boundaries

TABLE 2.22.3 The annual external costs of modern agriculture related to soil erosion in the UK and Germany (£ based on 1996)

Cost category	£ million	
	UK	Germany
I. Damage to natural capital: water		
a. Pesticides in sources of drinking water	120	58
b. Nitrate, phosphate and soil in sources of drinking water	71	+
b. Eutrophication, pollution incidents, fish deaths, monitoring costs	17	33
II. Damage to nature capital: soil		
a. Off-site damage caused by erosion	14	+
Flooding, blocked ditches and lost water storage		
Damage to industry, navigation and fisheries		
b. Organic matter and carbon dioxide losses from soil	82	+
Total annual external costs related to soil erosion	**304**	**>91**

Source: Pretty *et al*. (2001).

of the system that have to be taken into account. McConnell (1983) and Ervin and Ervin (1982) pointed out that in addition damage caused by flooding or damage to housing, water pollution is an external effect of soil erosion that might even exceed other effects. Therefore, the following considerations include direct soil erosion problems and indirect water pollution problems that can be assigned to soil erosion.

Research on the quantification of external effects of soil erosion is most advanced in the USA and Australia. Only few a examples can be shown for Europe. Colombo *et al*. (2003) calculated costs of soil erosion in a Spanish watershed of around €42–72 $ha^{-1}\,yr^{-1}$ using a contingent valuation method. Pretty *et al*. (2001) quantified the total annual external costs of agricultural production in the UK as £2342 million (€ 3465 million) and in Germany as at least £1230 million (€1820 million) at 1996 prices (see also Boardman *et al*., 2003). However, soil erosion only contributes a fraction to this amount, as documented in Table 2.22.3.

Table 2.22.3 clarifies that, especially outside the UK, there is a lack of investigations assessing external costs of soil erosion. The character '+' demonstrates, that with the utmost probability, there is an effect. The character '>' in the last row demonstrates that the measured effect in Germany probably underestimates the real effect considerably.

2.22.2.2.3 Approaches to Assessing the Value of Soil Conservation by Assessing the Benefit for Society

In addition to the cost-oriented approaches discussed in the previous section, socio-economists have developed methods to evaluate the benefit that society assigns to resource conservation. These approaches trace back to the theory of public goods that are not tradable on markets. Public goods are characterised by two decisive issues (Musgrave and Musgrave, 1989). First, contrary to private goods, there is no competition concerning the benefit of the use of public goods i.e. the benefit of such a good for individual A by using it does not affect the benefit of this good for individual B, who can also use it. A typical example is the pleasure people feel on contemplating an appealing landscape or by listening to a singing bird. Second, there are no mechanisms to exclude potential users from using this good. The given example also illustrates this issue. It is important to state that there is a smooth transition between private and public goods. Therefore, economists prefer to talk about goods with a high or low level of publicity. The theory of public goods is closely connected with the theory of external effects discussed above.

The theory of public goods show that in the case of natural resources, there are other benefits than those expressed by market regulations or market values. Table 2.22.2 gave a brief survey of possible values that are represented by public goods. Within the social and economic sciences, methodologies that allow for the investigation of such topics are available in principle. Such approaches vary from economic methods to approaches that are derived from empirical social research. Well-elaborated and well-known approaches include contingent valuation and the analysis of willingness to pay. In the context of this chapter, it is not possible to explore this topic in detailed way, but comprehensive surveys are given by, e.g., Pearce (1993), Willis and Corkindale (1995) and Bateman and Willis (1999). For critical assessment of such methods, see, e.g., Hausman (1993) or Hampicke (2003).

However, to date no investigations are available that assess the primary value, the intrinsic value or bequest value of soil in an explicit way. Anyhow, public authorities allocate subsidies for soil conservation programmes. It can be argued that intervention of the public authorities expresses the willingness of society to allocate money to maintain the values of soils, that are not a matter of private market activities and meet the conditions of public goods described above. Therefore, society's willingness to pay for soil conservation in terms of public goods may be estimated approximately by adding up public expenditure in this domain.

In order to analyse the willingness to pay through public investments, we focus on EU regulations. This means that the following reflections describe the EU share of public expenditure. National programmes may add to it. The 5th and 6th Environmental Action Programmes (European Commission, 2001a and 2001b) rules the integration of environmental protection into other sectors such as the Common Agricultural Policy (CAP) in general. Since 1999, the effort of the CAP to protect natural resources has been arranged in its second pillar. Regional agro-environmental programmes are the preferred instruments for implementing nature conservation issues. In 1992, the first regulation (2078/92) covering all EU Member States was passed. In 1999, new schemes were submitted and the old regulation 2078/92 was replaced by the current regulation 1257/99 (Baldock and Bennett, 2002). The effects of these programmes for reaching nature goals is, *inter alia*, the subject of the mid-term review of agro-environmental policy that is currently being carried out. Another aim of the mid-term review is to provide a structured survey of the very diffuse situation concerning agri-environmental programmes within the EU Member States or regions. As up to now there are no data available that summarise all subsidies distributed for soil conservation measures all over Europe, some rough calculations that are based on example and plausibility considerations may serve alternatively.

The state of Brandenburg in Germany distributes €40.65 million per annum within the scope of regulations 2078/92 and 1257/99 (MLUR, 2003). The proportion of soil conservation measures within this programme amounted in 2002 to €3.4 million yr^{-1}, which is approximately 8.3 %. In Germany, the total volume of agro-environmental programmes amounted in the same year to €1.995 million. Supposing that the share of soil conservation measures corresponds approximately to the situation in Brandenburg, the total expenditure for soil conservation measures in Germany that are allocated by means of agro-environmental programmes sums to €166 million.

Referred to the value of the whole agricultural production in Germany in 2002, amounting to €41 587 million, soil conservation measures have a share of 0.4 %. This result may be interpreted from two different sides. First, the relevance that society attributes to soil besides its value for primary production at present is almost negligible. However, it can be stated that society is willing to spend scarce financial means for soil maintenance and thereby expressing its willingness to pay for soil values beyond the mere production function.

The comparison with on-farm effects and costs for negative external effect indicates that public expenditure for soil conservation measures should not be underestimated. As the total annual costs for private on-farm effects are estimated at £8.65 million (€12.8 million) for England and Wales (Evans, 1995), and total off-site costs are estimated at £304 million (€450 million) in the UK and at least £91 million (€135 million) in Germany (Pretty *et al*., 2001), public expenditure amounting to approximately €166 million (£112 million) for soil conservation measures in Germany constitutes a remarkable dimension.

2.22.3 ASSESSMENT OF SOIL EROSION BY MEANS OF INSTITUTIONAL AND POLITICAL ECONOMY

It is a crucial point to recognise that soil erosion addresses different stakeholders with differing interests and demands involved. The resulting legal handling of soil erosion including thresholds and acceptance levels is based on a societal agreement which has its expression in property rights, that is, the right to cause damage or the right of not being the victim of damage, laws or other regulations. For general definitions of property rights, see Coase (1937), Bromley (1989), Scott (1989a,b) and Challen (2000). Within the process of negotiating about property rights, transaction costs play an important role in whether or not the actors start this process at all. These transaction costs influence the preferability of certain measures or policy instruments. Transaction costs have to be added to the actual costs, emerging both for the farmers and the participants involved such as governmental agencies (Williamson, 1985; Falconer, 2000; Falconer *et al*., 2001). In the following, these effects of soil erosion from the viewpoint of institutional and political economics are discussed.

2.22.3.1 Property Rights – Who has the Right to Pollute?

Regarding the use of the limited resource 'land or soil' for the production of food and fibre, a certain level of pollution is usually accepted by society for reasons of economic growth and welfare. What level of erosion damage should be normatively accepted by society (i.e. a threshold for erosion damage) requires government interference. Given the complex nature of erosion processes, thresholds are hard to define and become meaningful only in relation to some probability considerations.

The analysis of property rights can be used to find out if a rearrangement of those rights could lead to a better solution of environmental problems such as off-farm damage. According to Coase (1937), the optimal solution of such a problem does not always mean avoiding this damage completely. If property rights are assigned between the producer of damage and the victim of damage in a well-defined way, both can negotiate about compensation. Therefore, the detailed and precise distribution of property rights to certain goods can help to avoid legal conflicts. Property rights not only mean the right of possession, but also comprise the rights of use or destruction of a certain good (Bromley, 1989). If the farmer has a right to produce on his fields, but no right to pollute public or private property other than his own, it is still a question of costs whether the farmer will choose to avoid erosion or pay the fee for the clean-up instead.

There is a wide range in which to distribute possible sets of property rights in the case of soil conservation targets. If property rights are defining a practice that does not allow any pollution, farmers are obliged to use production practices that do not cause any soil erosion at all. The agricultural production in this case is neutral in terms of externalities. Assuming that society has consented to accept a certain amount of erosion, property rights are defined in such a way that society is willing to accept negative externalities, e.g. in order to have a higher supply of food in the short run. This point can be described as a set of property rights that give a farmer a right to pollute up to a certain limit or above a certain threshold of external conditions (e.g. rainfall intensity). In the case of soil conservation, this could be compared with a level of 'good agricultural practice' that does not avoid soil erosion completely, but sets limits on practices that would cause a high risk of erosion events.

A farmer practising in the EU according to 'good agricultural practice' is usually well provided with the rights to produce in a way that is approved by society. It is important to state that in the case of an agricultural policy based on the paradigm of 'good agricultural practice', agricultural activities are still based on a right to pollute although they can be seen as a negative externality. The concept of property rights provides an explanation of why soil erosion is still a common problem throughout the EU. Within the EU, land users have, at least to a certain extent, the right to pollute the environment with sediment originating from their soils. If

society's attitudes towards these externalities change together with changing scarcities of food or soil and water quality, society has to initiate for a change in farmers' behaviour.

2.22.3.2 Different Options for Changing Land Use Towards Soil Conservation in the Context of Property Rights

Any change in farmers' behaviour can either be achieved by a change of property rights, below termed Path A, or within the present set of property rights by financial incentives, below termed Path B. One possibility for changing farmers' behaviour through changing property rights is to revise the guidelines of 'good agricultural practice', or a more drastic change of legal boundaries has to be undertaken, which is usually a time-consuming task and can cause severe resistance from the stakeholders involved. Both approaches reflect on a change in property rights that results in an explicit restriction of farmer's rights. In the following, these approaches will be discussed as Path A(I) and Path A(II). A more voluntary way for society to avoid soil erosion damage is to set financial incentives to make certain soil conservation measures economically more or less preferable (e.g. subsidies for soil conservation tillage). Finally, there is still one more option for handling effects of soil erosion. Society may cover the claims of people who suffer through off-farm damage. In the latter case, property rights stay untouched, i.e. society accepts a strong position of farmers and seeks compensation either for the farmers or for the injured party. In the following, these approaches will be discussed as Path B(I) and Path B(II).

The possible distribution of property rights and the options to change them are described in Table 2.22.4. In the following sections, the implications of these options will be examined.

TABLE 2.22.4 Possible distribution of property rights and options to change land use

State of property rights on soils	Options to change land use	Comments
Path A: Changing property rights of the landowner	(I) Buying the right to pollute from the landowner in the form of lump sum compensation	If any damage occurs neighbours can claim from the landowner for compensation
	(II) Changing the legal framework to prohibit any kind of off-site damage without compensation	This option will face high resistance among landowners, since it is partly a dispossession of landowners
Path B: Property rights stay unchanged	(I) Society sets financial incentives to landowners/users to promote a voluntary less risky land use	Difficult to adjust the right amount of incentive: if it is too small, no participation; too high, too many free riders and windfall gains. Also, if it is a fixed payment, per hectare it does not consider different individual prevention costs.
	(II) Society covers claims of neighbours who suffered off-site damage	Complete inversion of the 'polluter pays' principle. Usually not easily accepted by society. However, given high transaction costs, can be a more efficient solution than others

2.22.3.2.1 *Path A (I). Buying the Right to Pollute from the Landowner in the form of a Lump Sum Compensation*

Given a high interest by society in a change of land use and property rights that cannot be changed by legal process or only under the burden of high transaction costs, e.g. given a high, powerful resistance by landowners being organised in strong interest groups, the following option will be favourable. Landowners would receive a lump sum compensation for a restriction of their property rights. If any damage occurs, neighbours can claim compensation from the landowner, since the landowner is now responsible for keeping any harm away from their neighbours.

2.22.3.2.2 *Path A (II). Changing the Legal Framework to Prohibit Any Kind of Off-farm Damage Without Compensation*

This option will face high resistance among landowners, since it means partly expropriation of landowners. Nevertheless, it can be justified with the obligation to avoid any damage to others while using a certain good. Assuming a society with sparsely organised landowners with little power, but powerful groups of environmentalists and other potential claimants, this option will be preferable, though, since the process of building such a legal framework will face only low transaction costs. Additionally, society will not have to pay compensation to farmers and claimants. In the case of soil conservation, an example would be a very restrictive formulation of 'good agricultural practice', comprising, e.g, obligatory reduced tillage systems.

This solution often seems to be the best solution when it comes to environmental problems, not taking into account that the overall costs of such a procedure are higher than just the enactment costs. The compliance with given laws must be feasible with a reasonable budget, otherwise the benefits of such a law are easily outweighed by the administration costs.

2.22.3.2.3 *Path B (I). Society Offers Financial Incentives to Landowners/Users to Promote a Voluntary Less Risky Land Use (Production of Non-commodity outputs)*

By setting up soil conservation programmes, policy makers try to set financial incentives to promote more soil-conserving agricultural practices. The concept of multifunctionality of agriculture as described in Wiggering *et al*. (2003) can be applied in this context.

The prevention of soil erosion above a certain level could be defined as the production of a non-commodity output, which has a certain value to society. Hence it is justifiable to pay farmers a price for this product. Non-commodity outputs are defined as products not being traded on classical markets for agricultural goods. Usually, these outputs depend on a demand expressed by governmental agencies through agricultural programmes. However, this demand can also be expressed by environmental organisations or groups of individuals damaged by soil erosion. The implementation of these agri-environmental policies brings up more questions about which instrument is either more effective or more efficient than another.

Governmental agencies have to decide which policy should be promoted, whether that be more extension services or the financial support of certain cropping systems. Furthermore, they have to decide whether the programmes should be targeted to certain sensitive areas or if the programmes apply to any farmer within a country. Whatever decision is chosen, it has to be taken into account that each choice brings budgetary costs with it, that do not only consist solely of the money paid to the farmer. There are also administration costs, i.e. the programmes have to be implemented and controlled and fraud has to be prosecuted.

It is of great importance to fix the right amount of incentive: if it is too low, there is no participation at all; if it is too high, windfall profits will likely occur, since farmers with low abatement costs who would have participated at much lower subsidies will join the programmes. Incentives based on a fixed payment per hectare do not take into account differing individual prevention costs.

2.22.3.2.3 Path B(II). Society Covers Claims of Neighbours Who Have Suffered Off-site Damage

This solution could be justified if the benefits originating from a certain type of agricultural land use are of public interest (external effects), which are higher than the private loss affecting neighbours. Although this land use might damage individuals, the result from the point of view of society might be better than a constrained land use. Since society gains from it, it is justifiable that the claimants are paid from public budgets.

This option could be seen as a complete inversion of the 'polluter pays' principle, which is usually not easily accepted by society. However, given high transaction costs, it can be a more efficient solution than others.

2.22.3.3 The Change of Property Rights Over Time

A simple example demonstrates how property rights can change over time. If a resource is not scarce, the likelihood of any argument about this resource is small, but the more a resource becomes limited, the more precisely a property right to this good will develop. Uncultivated wetlands or swamps along rivers were of low interest to ancient societies, because even agricultural use was very limited without any drainage. Therefore, at this stage of technology, property rights were not defined precisely. As a consequence, any damage caused by floods was not even noticed by society. Later, when technological developments permitted agricultural land use in the wetlands, property rights were assigned to the land users. Floods, at least to a certain extent, were welcomed by farmers, since sediment increased the productivity of their fields. However, already at this point, the action of one farmer could harm the property of others, e.g. through the level of water adjustments, which could flood other fields. Also, at this stage, institutional arrangements had to be developed. Some generations later, when house owners started to build houses in these lowlands, competing interests emerged. Whereas farmers can still accept floods to a certain extent, house owners tried to find ways to avoid those floods. The question arose of whether house owners, claiming property rights to the ground on which they build their houses, can also demand absolute protection against floods.

2.22.3.4 Transaction Costs Accompanying a New Setup of Property Rights

The change of property rights is usually accompanied by transaction costs (Coase, 1937). These costs are defined as costs of arranging a contract *ex ante* and monitoring and enforcing it *ex post* (Matthews, 1986). They can also arise from negotiations about a complete rearrangement of property rights. Moreover, the design, implementation and control of environmental programmes can cause transaction costs.

The costs occurring can be assigned to different types, derived from the purpose from which the costs arise. Thompson (1998) developed a checklist for those types accruing on different stages of the implementation of policies (Table 2.22.5). This list can also be used to analyse a change in property rights, since a governmental intervention means a change of property rights.

The compliance costs of a certain policy are the only type of costs which are directly allocated to the farmer. They can be estimated on the basis of additional effort in labour time (on field, applying for programmes, information gathering), additional expenses for machinery and, e.g., seeds and the decrease in yield caused by more extensive cropping systems (see Kalettka *et al.*, 2001, for an example). They can also increase when a new law forces farmers to adopt or cancel certain measures on their fields.

The enactment costs can be taken as a lump sum, since there will not be more Members of Parliament because of a change in policies or property rights. The same might be true for lobby groups. Nevertheless, both groups might spend less time on other topics that might be more useful.

The implementation costs may vary greatly depending on the precision and site specificity of the chosen instruments. However, it is hard to measure these costs. Detection and prosecution costs depend on whether a certain action (conservation tillage) or a result (e.g. soil loss in tons) is awarded or just prohibited by law.

TABLE 2.22.5 Different cost types within the institutional transaction costs (ITC) framework

Cost type	Explanation
Compliance costs	E.g. how much do farmers have to pay for compliance for a certain programme or a new set of property rights (time, machines, information)
Enactment costs	Time and resources spent by legislators and lobby groups accompanying the enactment process
Implementation costs	Implementation of an alternative may involve determination of both its goals and means by an administrative agency
Detection costs	Does everybody really comply with a given law or policy?
Prosecution costs	Prosecution costs arise after an individual is detected as not being in compliance with a law or policy option

Source: Thompson, 1998. Reproduced with permission of DB Thompson.

It is important to realise that any measure or instrument used for the prevention of soil loss or off-farm damage bears costs that are assessed and anticipated by all the stakeholders involved. If the measures are to be implemented voluntarily, these stakeholders will look carefully at the financial and physical efforts that come along with these measures.

2.22.4 CONCLUSION AND OUTLOOK

The chapter reflects the role of economic considerations in the debate on soil erosion effects. Both on-farm private costs and off-farm public costs were discussed and theoretical considerations concerning property rights and transaction costs were introduced.

First, on-farm effects were discussed. It was shown both from theoretical and empirical points of view that soil erosion has an inferior relevance for farmers' behaviour concerning soil conservation. The assumption that market mechanisms will lead to the integration of soil conservation measures into farmers' decision making is theoretically doubtful and could not be shown empirically. Therefore, the need for governmental intervention in on-farm effects can be justified if these effects harm the vital interests of society.

Off-farm effects cause damage on a larger scale. First, off-farm effects of soil erosion go far beyond off-site effects and the terms must not be used synonymously. Off-site effects can be estimated by means of the analysis of costs caused by negative external effects of soil erosion. To measure society's appreciation of soil conservation, a benefit-oriented approach was introduced. Benefits can be estimated by measuring the willingness to pay for soil protection measures. Society annually allocates considerable funds for soil conservation by means of agro-environmental programmes that can be interpreted as a benefit that society expects from soil conservation. However, funds for soil conservation only contribute a small fraction in comparison with the total volume of agricultural production.

In the second part, assessment of soil erosion in terms of institutional and political economy is carried out with the main focus on the theory of property rights and transaction costs. This chapter discussed possibilities of how arrangements must be changed and which options exist to influence farmers' behaviour towards the application of soil conservation measures. In general, two main paths are imaginable. The first path aims at a change of property rights, i.e. farmers have to bear restrictions in agricultural production. The second path accepts the present set of property rights and tries to induce soil conservation measures through financial incentives. The latter leads to an approach that even accepts present soil erosion levels and seeks possibilities that society pays for people who suffer from off-farm damage. From a scientific point of view, no best solution can be determined. A brief discussion of the chances and obstacles of the presented alternatives was given.

Finally, some considerations respecting the decisive role of transaction costs associated with policy change were made. It is important to realise that any measure or instrument used for the prevention of soil loss or off-site damage bears costs that have to be taken into account, in addition to the mere on-farm and off-site costs. Neglecting transaction costs leads to an underestimation of the real costs arising from soil conservation and presents a true perception of both obstacles and of possible solutions for the realisation of long-term soil preservation.

Even though numerous studies dealing with socio-economic effects of soil erosion are available, quantitative results are scarce. Especially for Europe, more research is needed on the damage caused by off-site and off-farm erosion effects, on the calculation of indirect and non-use values and on transaction costs accompanying conservation schemes and measures. Such investigations are essential for developing better targeted and designed policies related to soil conservation.

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2.23

Government and Agency Response to Soil Erosion Risk in Europe

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2.23.1 INTRODUCTION

'The dust is gold that bears the harvest;
Save the soil that grows our bread;
Let not wind and rain remove it;
Guard with care for years ahead.'

SG Brade-Birks (1944)

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Soil conservation in Europe has not generally received sufficient attention, until recently. However, a picture has gradually emerged, largely developed by individual research teams from various academic institutions. In Northern Europe, the Katholieke Universiteit Leuven (Belgium) has made a particular contribution. Erosion surveys were often conducted by Quaternary geologists and geomorphologists, who increasingly viewed evidence of agriculturally induced erosion in their landscapes (Fullen, 2000, 2003).

Faced with evidence of increased frequency and severity of soil erosion, a considered approach to soil conservation is essential. This must take into account both the common attributes and diversity of European agronomic, environmental and social conditions. The soil conservation policies of 10 European countries (Belgium, Denmark, France, Germany, Hungary, Iceland, Italy, The Netherlands, Spain and the UK) are reviewed. These countries were selected as representative of European regions (i.e. north, east, west and Mediterranean Europe, plus Scandinavia). The soil conservation policies are compared with several Australian and US strategies and several policy instruments recommended as appropriate approaches to soil conservation throughout Europe.

2.23.2 REVIEW OF EUROPEAN NATIONAL POLICIES

2.23.2.1 Belgium

Individual regions of Belgium are responsible for their own environmental issues; thus soil conservation policies differ between the Flanders and Wallonia regions. For instance, soil degradation has been recognised as an environmental problem for more than a decade in Flanders, resulting in the conception and implementation of the Flemish Environmental Action Plan (1997–2001). In contrast, policy developments have not yet been observed in Wallonia. Although several agri-environmental measures (e.g. cover crops and grass buffer strips along rivers and field boundaries) are implemented by many farmers experiencing erosion problems, none were originally meant to control erosion as part of regional policy (Bielders *et al*., 2003).

The latest Flemish Environmental Action Plan (2003–07) recognizes erosion as a threat to long-term soil sustainability and addresses soil protection through erosion control. In December 2001, the Flemish Government issued a decree concerning 'the subsidy of small-scale erosion control measures to be taken by local authorities', often called the 'Soil Erosion Decree'. This Decree regulates subsidies to municipalities on two levels. First, municipalities dealing with erosion problems in the hilly regions of Flanders are stimulated to draw up a municipal soil erosion action plan, indicating their proposed measures. The Flemish Government provides municipalities with €12.5 ha^{-1} of plan area. Once the Land Division of the Flemish Administration (AMINAL, Land) approves a plan, it grants a 75 % subsidy for the implementation of approved measures. The measures are selected from a 'Code of Good Practice – Erosion Control', currently being prepared by the Flemish Administration, and will include mostly small-scale technical control methods, such as the construction of small dams, sedimentation ponds, grass buffer strips and grassed waterways. Soil conservation measures carried out by farmers, such as conservation tillage or cover crops, are not subsidised via the 'Erosion Decree', but can be recommended in a municipal soil erosion action plan. However, for other environmental reasons, farmers can be subsidised annually for sowing cover crops during autumn (€50 ha^{-1}) and for sowing grass strips with a width of 5–10 m along rivers and sunken lanes (€0.13 m^{-2}). These practices are now also promoted within the framework of the soil erosion control policy. Other subsidies for farmers, relevant for soil erosion prevention, include subsidies for afforestation (between €850 ha^{-1} for poplar and €3700 ha^{-1} for oak) and set-aside (€298–424 ha^{-1}, depending on agricultural region and crop type). Further grants for agricultural practices which decrease soil erosion are expected to be available in 2004, through the 'Flemish Plan for Rural Development', within the framework of European Directives. These measures will

include soil conservation actions at the field scale and were approved by the European Commission (EC) in August 2003.

Establishment of a municipal soil erosion action plan occurs in close cooperation with farmers and this participatory approach has been applied recently in many Flemish demonstration projects (Verstraeten *et al*., 2003). Only when farmers are fully involved in project design and dissemination can new conservation techniques be successfully implemented on a broader scale. To assist municipalities in establishing a soil erosion action plan, the Flemish Government subsidises them for organising awareness raising actions such as erosion-prevention demonstration fields (€125 per field), for excursions with farmers to an erosion prevention project (€250), for advisory courses (€50 per session) and to approach individual farmers in order to put the proposed actions into practice (€50 per farmer).

Flemish policy makers use several soil erosion indicators (Vandekerckhove *et al*., 2003). The most important indicator is the soil erosion risk map of Flanders, based on the application of the RUSLE with 20-m resolution and aggregated at the field scale (Van Rompaey *et al*., 2000). Furthermore, long-term effects of soil erosion on soil fertility are evaluated for each field, using soil profile information from the 1:20 000 Belgian Soil Map. Effectiveness of conservation measures is estimated by modelling decreased soil loss at the field scale and decreased sediment delivery to rivers, using a spatially distributed soil erosion and sediment delivery model (Verstraeten *et al*., 2002). The Walloon government is keen to establish a soil erosion control policy soon, embracing the Flemish model.

2.23.2.2 Denmark

Denmark has low erosion rates compared with areas of southern Europe (European Environment Agency, 2000), yet there is a very active programme of soil conservation, particularly in terms of coastal wind erosion control (Als, 1989). This was initiated during the 19th century, when Hedeselskabet (the Danish Land Development Service) provided free plants to establish windbreaks to address serious erosion problems. Regional shelterbelt projects can cover 50 % of expenses for soil preparation, plants, planting and hedge maintenance for 3 years. Despite the high investment costs, this policy resulted in the planting of 1 000 km of shelterbelts per year (Veihe *et al.*, 2003). Today, Hedeselskabet continues to offer technical support for establishing windbreaks, but government subsidies are available for the purchase of certified plants. The success of these schemes is attributed to farmer participation and good products, plus government involvement, by actioning a specific windbreak law in 1976, which has been revised several times (Veihe *et al*., 2003).

Tillage erosion rates are relatively high in Denmark, at an estimated average of $\sim$6 t ha^{-1} (Heckrath, 2000). The Foulum Research Centre started a 3-year EU-funded project (FAIR3) in 1997, with the aim of quantifying tillage erosion, determining its effect on soil fertility and investigating the influence of farming practices on soil erosion and quality (Djurhuus and Heckrath, 2000). Yet there still remains no law dealing specifically with tillage erosion (Veihe *et al.*, 2003).

In 1986, the Danish Nature Council and the national media publicly highlighted eutrophication problems, which, supported by the Agricultural Council, forced the government to inaugurate the Water Environment Protection Plan I (WWPP I) and the NPo (nitrogen, phosphorus and organic matter) research programme. However, policy measures to address water erosion are initiated through the designation of 'Specifically Vulnerable Agricultural Areas' (SVAA). Government subsidy measures include set-aside, 2-m wide buffer strips around all watercourses in rural areas and use of rye-grass catch crops (Sibbesen and Iversen, 1997). WWPP I was superseded in 1998 by the Water Environment Protection Plan II (WWPP II), after erosion control became recognised as a mechanism for reducing agrochemical contamination.

It has been argued that the Danish Land Development Service (Hedeselskabet) could act as a general organisational model for Europe (Morgan and Rickson, 1990), as they operate an integrated approach to landscape management (Dubgaard, 2000). However, a change of government in late 2001, associated with

major environmental sector cut-backs, has been accompanied by an attempt to simplify laws to ameliorate working conditions and increase profit within the agricultural sector. This is instigated by the notion that farmers are best placed to formulate their own decisions regarding farming practices and concomitant environmental effects (Veihe *et al*., 2003).

2.23.2.3 France

Many areas of France have a long history of soil erosion problems. For instance, in mountainous areas a state organisation (Mountain Land Restoration of the National Office of Forests) has been responsible for soil protection for over a century (Lilin, 1986). Elsewhere, in the Bourgogne vineyards, 19th century private contracts forced farmers to retrieve and replace eroded soil (Durousset, 1994). In rural areas, no specific contracts or laws were applied to soil protection and a national survey in 1950 revealed soil erosion in mountainous areas had been alleviated and insignificant erosion problems existed on farmlands (Hénin and Gobillot, 1950). This situation prevailed until the 1970s.

The first French agricultural law (1960), preceding the first common agricultural policy (1962), led to conversion of grasslands to cereals and field enlargement (Vivier *et al.*, 1985). Increases in runoff and muddy floods have also been linked to post-War urbanisation. These storm events often had disastrous and sometimes fatal consequences and caused increased infrastructure damage. A proposed solution was a natural disaster law (1982), aimed at providing compensation for the victims of natural disasters, but this failed to address the source of the problem.

There continues to be no specific legislation on soil protection in France, because soil erosion is an unrecognised problem for and by farmers. Moreover, widespread chemical use or irrigation systems are considered to balance negative impacts of soil erosion on agricultural production. Instead of soil protection, water (quality and quantity) is viewed as the principal national environmental issue. However, water policies have some positive effects on erosion control. The Water Law (1992) initiated the division of France according to the six main catchment areas and established the foundations of the coordinated and sustainable development of water resources. The Water Law significantly increased the role and responsibilities of local authorities, reinforced regulations and provided new tools (Schéma Directeur d'Aménagement et de Gestion des Eaux, SDAGE). In this new framework, the designation of protected areas around catchments ensured the compulsory reintroduction of permanent grassland and helped local stakeholders manage problems at the catchment scale instead of at the water resource (i.e. river, lake) level. The framework considerably assists in dealing with erosion issues. At the same time, the Common Agricultural Policy (CAP) reforms (European Regulation No. 2078/92) introduced some positive agri-environmental measures for erosion control, mostly those concerning both protection and extension of grassland. Measures for extensification of livestock help to maintain grassland in highland areas. Other long-term (i.e. 20 years or longer) measures included set-aside of agricultural land (payment €457 ha^{-1}) and the reversion of arable land to extensive grassland (payment €381 ha^{-1}). However, in areas of intensive agriculture, these measures were unattractive. For instance, Upper Normandy gave a supplement of €76 ha^{-1} to encourage the reversion of arable land to extensive grassland for 5 years. In this erosion prevention instrument, grasslands must be located in thalwegs. Farmers received subsidies corresponding to the long-term set-aside of agricultural land without being required to commit themselves for 20 years. In 1999, the French farm framework law No. 99-574 introduced 'Territorial Exploitation Contracts' (CTEs) to establish rural development projects that addressed environmental, economic and social issues. Access to agri-environmental measures became conditional on agreeing to a CTE. Depending on local needs, some CTE have focused on erosion control, but difficulties in implementation and high cost led to the halt of CTE measures. In 2001, CTEs were replaced by a new contract ('Contrat d'Agriculture Durable', CAD), which is more economical since it only considers environmental issues.

Despite the lack of national soil protection legislation, local public authorities became aware that erosion control involves coordination (Cartier, 2002). Consequently, some authorities formulated plans to solve environmental problems, notably erosion. For example, recent erosion problems in Seine-Maritime led to the designation of 'Syndicats de bassin versants' (hereafter referred to as the Syndicates), corresponding to the main catchment areas. In accordance with the State–Region Plan, Water Authorities control the steering committee of Syndicates. Syndicates recruit advisors, with hydraulic or agricultural abilities, charged with implementing hydraulic solutions and encouraging farmers to change agricultural practices.

At the local level, it became evident that both hydraulic and agricultural solutions to soil erosion problems were necessary. However, modifying agricultural practices is difficult, for three reasons. First, farmers' decisions are strongly influenced by technological and commercial considerations and economic support under CAP, plus changes in crop management systems have fuelled environmental uncertainties (Souchère *et al*., 2003). Second, the diversity of natural conditions (e.g. climate, soils), crops and agricultural practices causes difficulties in establishing and adapting technical guidelines for agricultural practices suitable for these different conditions (Papy *et al*., 1996; Martin *et al*., 2004). Finally, since agricultural extension services are directly financed by farmers' taxes to advise farmers how to increase their net income, soil erosion was not viewed as a farming problem until recently. The main reason is that agricultural advisors experienced difficulties in estimating the efficiency of new cultivation techniques and also their economic consequences (Cattan and Mermet, 1998). Furthermore, the incentive for advisors to promote change fails when agricultural prices decline, because their own funding is reduced. In future, it is hoped that local communities may subsidise agricultural extension services to contribute to the development of adapted technical guidelines, in cooperation with the National Agronomy Research Institute (INRA).

2.23.2.4 Germany

In Germany, soil conservation legislation has recently been developed, with the main stress on conserving water quality (Weingarten and Frohberg, 2000). For instance, some soil conservation requirements were incorporated into the state law of the German Lande of Baden-Württemberg in 1991 (Jäger, 1994). Now, over 30 Federal laws and regulations govern environmental protection. The Federal Soil Protection Act (Act on Protection Against Harmful Changes to Soil and on Rehabilitation of Contaminated Sites) became Federal Law on 1 March 1999 (Frielinghaus, 2001). The purposes of the Act are to protect and restore soil functions on a permanent and sustainable basis and to concentrate on both precautions and hazard protection. The latter is achieved by the provision of a code of good management practice, dealing especially with risks of soil erosion and compaction. Good farming practice includes designing appropriate treatments with regard to the site and climatic conditions, maintaining and improving soil structure, promoting biological activity and protecting soil humus from deterioration. The Act provides a useful model for European soil protection legislation and is being consulted in the design of legal instruments in other European countries (e.g. Belgium and the UK).

2.23.2.5 Hungary

Soil erosion is a major soil degradation process in Hungary (Várallyay, 1989; Kertész 2001). Soil protection legislation includes many forms of soil degradation, such as soil erosion, salinity, extreme acidity and soil pollution. Before the political regime change of 1989, Hungarian agriculture was highly subsidised, making it one of the most successful branches of the national economy, with high crop yields and a good reputation, which was exceptional among the former socialist countries.

Soil conservation legislation is enshrined in Chapter VI of Hungarian State Law No. 55 (1994), in which soil conservation is considered a joint task between the State and the land user. The State develops national soil conservation policies, ascribes to international agreements and provides farmers and planners with information

systems on soil quality, in the form of maps, soil data and legal, economic and technical instruments. A soil conservation service has been initiated ('Service for Plant Protection and Soil Conservation'), comprised of a national network of plant health and soil conservation stations, with one in each of the 19 Hungarian counties.

Land users are obliged to conserve topsoil and its organic matter content and to conduct environmentally friendly farm management. This is achieved by appropriate land use, cultivation technology and soil conservation measures. Law 55 includes actions for eroded and erodible soils, including land-use regulations, provision of protective vegetal cover and cultivation methods. If these measures cannot guarantee good protection against erosion, a conservation plan must be formulated, specifying appropriate actions.

After the regime change, a complicated reprivatisation process started, resulting in a very complex land ownership structure. Some cooperative farms ceased to exist, while some remained with changes to their legal status and many new private farms were established. These were diverse, in terms of both size and financial viability. Consequently, most new farms are facing challenging economic positions, so soil conservation is given low priority, which is a recurrent problem in the other former socialist countries.

Law 55 stresses that both land users and the State are important partners in soil protection. It is imperative to learn farmers' opinions on soil erosion and conservation. According to a case study of 44 farmers in Keszthely, near Lake Balaton, two-thirds of farmers would not spend money on erosion control, even if they had the resources (Puskás, 2002). However, approximately the same proportion of questioned farmers were convinced about the damaging effect of erosion and 40 % of them considered soil loss due to erosion was very high. Some 50 % of the farmers did apply soil conservation measures and only 25 % remained passive and did not take any conservation action.

Joining the EU in 2004 will certainly change both agricultural policy and the attitude of farmers in all Accession States. The EU strategy on soil conservation will influence the national policies of these countries, hopefully in a positive direction.

2.23.2.6 Iceland

The severity of soil erosion in Iceland prompted the creation of Northern Europe's only designated soil conservation service, the Landgraedsla Rikisins (SCS), in 1907 (Runolfsson, 1978; Arnalds, 2000). The first 50 years were mainly devoted to halting sand dune encroachment and other forms of catastrophic soil erosion that were threatening several settlements. The Forestry Service, originally established by the same law as the SCS, had the role of combating woodland destruction and promoting reforestation. It is possible to identify several attributes of successful practice from the almost 100 years of combating ecosystem degeneration and desertification in Iceland.

About half of the vegetation and 95 % of woodland cover has been lost in the 1100 years of settlement, and the condition of the remaining vegetation is severely degraded in many areas. Serious soil erosion characterises 40 % of Iceland according to a national survey completed in 1997 (Arnalds *et al.*, 2001). This survey represents a major breakthrough, as increasingly, people accept the reality of erosion problems and focus on their solution.

Since 1990, there has been an increasing participatory approach to soil conservation, which has markedly increased the adoption and success of conservation projects (Arnalds, 1999). The 'Farmer's Reclaim the Land' Project (Arnalds, 2000) includes a 'cost-share' partnership with farmers, with conservation work jointly funded by government and farmers. This 'bottom-up' approach encourages involvement and individual ownership of conservation projects. The rapidly increasing forestry in Iceland, which also has a large role in land improvements, has a strong farmer and public participation focus.

Current developments include a Parliament-approved programme, which gives the SCS an operational framework for the period 2003–14, with increased funding. The main goals are mitigation of land degradation and desertification, revegetation of eroded land and attaining sustainable land use. Tools include increased

knowledge on problems and solutions, education and advice, land-user responsibility, legal improvements and widening participation. Carbon sequestration as a tool in meeting Iceland's obligations under the Kyoto Protocol has also become a major incentive for restoring land health (Arnalds, 2004).

Grazing by sheep and horses is a major determinant of land health in Iceland. In the heavily subsidised sheep industry (with $\geq$50 % of farmers' income being subsidised), the current contract between sheep producers and government has a cross-compliance clause that, starting in 2003–04, farmers must verify the ecological sustainability of their operation to the Landgraedsla Rikisins in order to obtain a full subsidy (Arnalds and Barkarson, 2003). In the important horse production sector, a voluntary 'bottom-up' quality control of sustainability is emerging. A new cooperative programme of farm planning, 'Better farms' combines the forces of soil conservation, forestry, extension and nature conservation in aiding land users in producing their own property plans. This is a very promising programme, which not only brings farmers into the planning process, but also improves coordination between the various institutions and organisations working with farmers. Icelandic soil conservation systems have been presented as an organisational model generally within Europe (Morgan and Rickson, 1990; Boardman *et al*., 2003).

2.23.2.7 Italy

Soil erosion plays an important role in Italian land degradation and the European Soil Bureau estimates that $\sim$70 % is at risk of accelerated erosion, defined as erosion $>5\,\mathrm{t\,ha^{-1}\,yr^{-1}}$ (Grimm *et al*., 2002). These risks relate to the widespread presence of steep erodible slopes, combined with intensive cultivation and lack of conservation measures.

National legislation considers soil conservation the result of public actions for soil stability and hydraulic security applied to soil, subsoil, water, urban areas and infrastructures. Italian Law 183 (18 May 1989) on 'soil defence' identifies a hydrological basin as the operating context for soil conservation planning. Each hydrological basin is defined by the law as a 'unitary ecosystem', i.e. a complex environment with its own homogeneity, where coordination and harmonisation of several functions related to soil conservation can be implemented, along with water management.

Establishment of the 'Basin Authorities', at national, regional and interregional levels, is one of the major innovations brought about by soil conservation reform. They assume responsibility for the territorial coordination of all the functions carried out by the State, regions and provinces with reference to matters listed in the law and this is accomplished mainly through 'Basin plans'. These include responsibility for soil stabilisation by both agrarian and forest hydraulic conservation structures, protection and consolidation of slopes against landslides and avalanches and prevention of both soil subsidence and seawater intrusion into aquifers and rivers.

Significant progress was made in soil conservation in 1999, when the 'Code of good agricultural policy' was approved by the Ministry of Agricultural and Forestry Policies. This contains national standards and directions that can be incorporated by regions and provinces to suit local requirements. The Code was approved under Article 4 of Directive 91/676/EC of the Council (12 December 1991) and implemented EC Regulation 1257/99. This regulation calls for agri-environmental measures to finance those actions extending beyond normal 'Good Agricultural Practice' (GAPn). Regions have defined GAPn to suit different local requirements and have developed detailed plans to decrease soil erosion risk (particularly decreasing soil tillage and mouldboard ploughing and promoting crop rotations that ensure soil cover during rainy seasons). As a result, farmers implementing CAP agri-environmental measures are given subsidies.

Other important legislation tools for soil conservation are Law 97 (1994) and the National Programme to Combat Drought and Desertification (NAP) (1999). Law 97 determines conditions necessary to enhance soil protection in mountain areas, through the development of farming activities that avoid soil erosion and maintain the landscape and natural resources. NAP encourages Regions and Basin Authorities to identify

'areas vulnerable to desertification' and 'remedial criteria', which includes the prevention and remediation of soil erosion and salinisation, slope protection through low-impact environmental actions, fire prevention and the adoption of farming, livestock and forestry management systems aimed at preventing the physical, chemical and biological degradation of soil.

Of the €146.26 million committed to implement NAP, 30 % is devoted to soil protection actions, 15 % to reducing the impact of industrial activities and 20 % to restoring soil equilibrium. NAP measures will continue to be adopted through existing funding instruments, such as rural development, water protection and river basin management plans, forest and afforestation programmes, EU Structural funds (2000–06) and CAP measures.

2.23.2.8 The Netherlands

During recent decades, soil protection legislation has primarily focused on prevention and remediation of soil contamination. The National Soil Protection Act (1987) provided the framework for establishing rules relating to categories of activities hazardous to soil. In addition, there are provincial environmental policy plans devoted to the protection of potable groundwater. Consequently, areas have been designated around extraction points within which activities hazardous to the soil require particular care. Furthermore, local authority environmental policy plans are in force and address specific soil protection issues. Additionally, agreements have been defined and formalised with several branches of industry, including agriculture, to limit the burden on the environment, and soil in particular. For instance, some soil conservation requirements have been incorporated by a ruling from the Dutch Agricultural Board (Landbouwschap) in 1992 (Boardman *et al*., 1994). Soil conservation organisational structures are developing in South Limburg, where a coordinated soil conservation project was conducted in the 1990s (Duijsings, 1994). The Limburg Soil Conservation Project (Erosienormeringsprojekt) involves active collaboration between government (provincial and municipality), agricultural advisory services, university research institutes and local farmers. Demonstration projects and information dissemination are important components of the programme (Boardman *et al.*, 1994; de Roo *et al.*, 1995). The soil conservation approach followed in the Limburg Soil Conservation Project has been applied elsewhere in The Netherlands, for instance in defining best conservation strategies for the erosion-prone area of Groesbeek in the eastern Netherlands (Stolte and Ritsema, 2001; Stolte *et al.*, 2002). Both the Limburg and Groesbeek Projects may well act as future models for European soil conservation policy.

Various national Ministries and related organisations have recently acknowledged that soil legislation and regulations are too narrowly focused and not harmonised with existing national and international environmental acts. Hence legislation should be reviewed, to make it more transparent and acceptable to the public, and more easily applicable and manageable for the respective authorities. In 2002, the Ministry of Housing, Spatial Planning and the Environment (VROM) established a Steering Committee 'Soil' (Stubo), in which other Ministries, provinces and municipalities participate in preparing a National Integrated Soil Policy Framework. This new policy should address soil protection in the widest sense, integrating the entire spectrum of soil physical, chemical and biological functions and properties. Through this process, soil conservation and runoff and erosion processes may receive appropriate governmental and agency responses.

2.23.2.9 Spain

Erosion has long been recognised in Spain (Mallada, 1890), but targeted research did not start until the second half of the 20th century (see Chapter 1.26), with the creation of the 'Servicio Central de Conservación de Suelos' (Central Service of Soil Conservation) in 1955. However, since adoption of the 1978 Constitution, Spain has become less centralised, comprising of 17 Autonomous Regional Governments with many responsibilities, including erosion control. Many soil erosion and conservation issues are administered by

different institutions within the Ministry of the Environment (Dirección General de Conservación de la Naturaleza, DGCN), formerly ICONA, and the 10 Basin Authorities (Confederaciones Hidrográficas).

Autonomous governments and national research institutions [e.g. CSIC, 'Consejo Superior de Investigaciones Científicas' (High Research Council for Scientific Research), and CEDEX ('Centro de Estudios y Experimentación de Obras Públicas', Centre for Experimental Studies of Public Works)] and universities have been the primary contributors to erosion research over the two last decades. In 1981, ICONA, in collaboration with CSIC and several universities, established the ongoing LUCDEME (Lucha contra la desertificación en el Mediterráneo) Project, to combat desertification in Mediterranean basins. Since then, maps of actual and potential soil erosion have been produced. Derived from LUCDEME, in 1995, RESEL [Red de Estaciones de Evaluación y Seguimiento de la Erosión y la Desertificación en España (Network of Experimental Stations for Assessing and Monitoring Soil Erosion and Desertification in Spain)] was established. The Project investigates erosion in problematic environments using field sites, at plot, hillslope and small catchment scales (Rojo and Sanchez-Fuster, 1997).

The National Research and Development Plan is the basic instrument for Spanish research policy and includes two National Programmes dealing directly with the causes, consequences and control of both desertification and soil erosion. For the last 20 years, soil erosion research has been conducted and funded by both national and European institutions. The EC has also given priority to large erosion research projects, such as DESERTLINKS, MEDACTION, MEDALUS, GEORANGE and PESERA.

Spanish agricultural policy, since joining the EC, has been dominated by the CAP, whose priority is the development of sustainable and ecologically sound agriculture. However, CAP implementation in southern Mediterranean European countries, and particularly in Spain, has had contradictory effects on soil erosion (e.g. set-aside resulted in a dramatic increase in the extent of bare soils vulnerable to erosion). CAP reforms, in the context of Agenda 2000, signified a new era of integrated development of rural areas, including soil protection and erosion control. Satisfying the new goals of the CAP, Agenda 2000 proposed related changes in the market-oriented agricultural and rural development policies.

A major development of compensatory payments is an option for countries to make direct payments, subject to compliance with environmental standards (EC1259/99). The Spanish eco-conditionality principle, adopted by Royal Decree 1322/2002, includes soil conservation practices for protecting fallow land from erosion, avoidance of stubble burning and prohibition of tillage on steep slopes.

Rural Development Policy Regulation 1257/99 provides funding from the European Agricultural Guidance and Guarantee Fund (EAGGF) to establish new development rural plans for Good Farming Practices (GFP). The Code of Good Farming Practices was approved by National Decree 4/2001 (13 January 2001). It contains valid national standards for tillage, alternate crops and the use of fertilisers among other aspects. These standards could be modified by regional governments to suit local requirements. Application of agri-environmental measures, regulated by National Decrees (4/2001 and 708/2002), pursue programmes of environmentally friendly agricultural soil protection practices. However, until recently, lack of coordination and co-finances meant bureaucratic delays in these programmes. Actual funded agri-environmental measures (€378.26 million over the period 1995–99) were less than the estimated mean annual cost of €258.4 million [Ministerio de Agricultura, Pesca y Alimentación (MAPA), 2000]. Since then, public investment of €1207 million (with an EU contribution of €827 million) million) has been committed for funding agri-environmental measures for 2000–06. Accompanying CAP measures are also scheduled and national funding committed.

A further instrument of policy development is the afforestation of agricultural lands, regulated by Royal Decree 6/2001, which substitutes former legislation of Regulation 2079/92, and establishes an Aid Programme for the afforestation of former agricultural lands. Its main objective is to increase the income of farmers affected by EU policies, by promoting forestry as an alternative use, to enhance long-term forest resources and to combat soil erosion and desertification (Gomez-Jover, 1996). In this case, programme implementation is the

responsibility of the Autonomous Communities, but is also co-financed by the central administration and the EU (EAGFF).

Since its creation in 1901, the Hydrology and Forest Divisions have been responsible for Spanish forest administration and have considerable experience in protecting soil against water erosion and rehabilitating degraded vegetal cover. Since their creation, most responsibility for forest resources and nature conservation has been transferred to autonomous communities. However, central government [through the General Directorate for Nature Conservation (DGCN) of the Ministry of the Environment] coordinates plans and programmes related to soil protection and desertification control. This includes the Spanish Forestry Strategy (EFE), the National Forest Plan (NFP) and the Autonomous Forest Plan. Objectives include the design of new forestry policies, emphasising multiple use of forests; organisation of forest resource responsibilities within the central administration, coordination with autonomous administrations and renewal of forest activity as a means of generating employment and economic activity.

The Forestry Strategy, developed by the NFP, promotes reforestation of degraded plant cover areas, wildfire protection and application of silvicultural treatments to improve forest quality [Ministerio de Medio Ambiente (MIMAM), 2001]. The Strategy foresees the reforestation of 3.8×10^6 ha affected by soil losses >5 t ha^{-1} yr^{-1} and the construction of 6.9×10^6 m^3 of control structures to stabilize torrential water courses. To address the application of programmes and actions to control soil erosion and combat desertification the NFP includes the 'National Plan of Prioritized Actions to Hydrological and Forest Restoration, Soil Erosion Control and Combating Desertification'.

Thirteen Autonomous Communities have drafted documents on forest policy and forestry master plans, of which seven have promulgated regional forest resource laws. Most plans identify the restoration of degraded lands affected by soil erosion, by the means of revegetation, and the need for sustainable forestry to control soil erosion and its off-site effects. For instance, the Forest Law of Valencia (Ley Forestal de la Comunidad Valenciana 3/93) highlights the fight against desertification, prevention of forest fires, restoration of vegetable cover and flood protection as major objectives. Thus, a reforestation plan was implemented for 1994–99. The plan foresaw re-vegetation of 100,000 ha of public lands, with a budget of €160 million.

The 'National Plan of Prioritised Actions to Hydrological and Forest Restoration, Soil Erosion Control and Combating Desertification' promoted by DGCN in 1991 (Rojo, 1998) aims to control soil erosion and establish locations and priorities for land rehabilitation in degraded river basins, according to both the use of river basins as operational units and soil erosion severity. Hydrological and forest restoration activities developed under 'Cooperation Conventions with Autonomous Government' (1997–2001) shows that the Plan is failing its initial goals.

In the future, 'The National Plan of Hydrological Forest Restoration and Erosion Control' will be integrated in the National Actions Programme to Combat Desertification (NAPD). When Spain became a signatory to the UN Convention to Combat Desertification (CCD) in 1996, it was committed to establishing an NAPD. This is being prepared by a working party group coordinated by DGCN, with the main objective of sustainable development in Spain's arid, semi-arid and dry subhumid areas and, particularly, the prevention and reduction of land degradation, the rehabilitation of partly degraded land and reclamation of desertified areas.

In 2002, DGCN initiated a national inventory of soil erosion (INES), which aims to locate, quantify and analyse the evolution of erosion processes in Spain. The main objectives are to prioritise target areas for soil conservation and define and evaluate national plans on reforestation, plant cover improvement and the management of forest biodiversity. In every province an inventory and map (1:50 000 scale) will be made of rill, gully, river bank and wind erosion and mass movements. Inventory maps have been completed for the provinces of Madrid, Murcia and Lugo, five more are at an advanced stage and another five are in progress. The inventory will cover all 50 provinces and will take about 10 years to complete. The inventory will then be repeated on a cyclical basis.

2.23.2.10 The UK

There is increasing evidence in the UK that soil erosion poses at least a moderate to severe problem at local to regional scales (Boardman and Evans, 1994), although government involvement remains only moderate. In the 1980s, the Agricultural Development and Advisory Service (ADAS), then a constituent part of the Ministry of Agriculture, Fisheries and Food (MAFF), published two advisory bulletins to help farmers minimise erosion (ADAS, 1984, 1985). Since then, a 'Code of Good Agricultural Practice for the Protection of Soil' has also been published (MAFF, 1993) and a detailed guide to soil conservation for farmers (MAFF, 1999). The Environment Agency (formerly the National Rivers Authority) was established on 1 April 1996, with a remit to protect the UK environment, including its soils. It is likely the Environment Agency will play a significant role in soil conservation (Boardman, 2002).

Several relatively simple approaches would reduce national soil erosion. MAFF recognised that set-aside schemes may contribute to soil conservation (MAFF, 1991, 1994, 1999; Margach, 1993) and, furthermore, grassland protection of riparian zones would decrease sediment delivery to streams and rivers (Morgan, 1992). The MAFF (1999) booklets advised farmers on simple techniques to produce a farm-size soil erosion risk map and approaches to combat erosion on areas 'at risk'. Particular emphasis was placed on grass strips on arable slopes and buffer strips in riparian zones. Grass strips were recommended to be 5–15 m wide and every 50–150 m downslope, with the width of the strip increasing and the gap between strips decreasing as erosion risk increased.

An independent, government-sponsored report advised the government to devise and implement a soil protection policy (Royal Commission, 1996). An outcome of the report was the preparation of a 'Draft Soil Strategy for England – a Consultation Paper' in March 2001 [Department of the Environment, Transport and the Regions (DETR), 2001]. Following public consultation, in which interested members of the public were invited to comment on the draft, the Strategy is being prepared for approval by the UK Parliament. By comparison, the nature and scope of the Strategy is similar to the German 'Federal Soil Protection Act (1999)'. In the more federal government structure, recently adopted in the UK, the legislation will be specifically for England, although legislation will be forwarded to the separate regional assemblies of Scotland, Wales and Northern Ireland. Although this will facilitate legislation fine-tuned to local conditions, it will inevitably slow the process of implementation, as fundamentally similar legislation will be subject to ratification by four separate legislative assemblies. The draft strategy is available at: http://www.defra.gov.uk/environment/consult/dss/

Progress has been made, with the submission of two reports. Stirling University (Scotland) reported to the Scottish Executive, recommending a soil protection strategy for Scotland (Adderley *et al*., 2001). The Centre for Ecology and Hydrology (Bangor, Wales) has presented a similar strategy to the Welsh Assembly. The respective texts are available at http://www.envsci.stir.ac.uk/spstrategy/index.htm and http://www.bangor.ceh.ac.uk/English/science/reports.htm

Agri-environment schemes in the UK, directed by MAFF [renamed the Department for Environment, Food and Rural Affairs (DEFR) in 2001], aim to secure environmental benefits above those of Good Farming Practice and cross-compliance conditions. Introduced in 1987 to implement EU Council Regulation 797/85, they were designed to prevent loss of habitat and landscape features associated with intensification at sites targeted by the Environmentally Sensitive Areas (ESA) Scheme. Subsequently, in 1991, the Countryside Steward Scheme (CSS) was established to provide incentives to landowners, farmers and other land managers to take specific measures to conserve, enhance and/or recreate important landscape types. In 1994, the Habitat Scheme (HS) was initiated to create, protect and enhance wildlife habitats by removing land from agricultural production and promoting environmentally sound land management practices. In 1995, the Moorland Scheme (MS) was launched with the objective of protecting and improving the upland moorland environment. In 1998, the Arable Stewardship Pilot

Scheme (ASPS) was created to assess alternative arable management options for conserving and enhancing farmland biodiversity (Ecoscope Applied Ecologists, 2003). In December 2003, the UK government initiated a new agri-environment initiative, known as the 'Environmental Stewardship Scheme' (ESS). The scheme operates two-tier participation. The 'Lower Entry Level Scheme' encourages farmers to deliver simple, yet effective, environmental land management, which will help to diffuse pollution, prevent loss of biodiversity and landscape character and damage to the historic environment. The 'Higher Level Scheme', launched in 2005, is based on the existing CSS and ESAs. Its five main objectives are wildlife conservation; protection of the historic environment; maintenance and enhancement of landscape quality and character; promoting public access; and understanding and resource protection (http://www.defra.gov.uk/erdp/reviews/agrienv/default.htm).

DEFRA is drafting 'The First Soil Action Plan for England 2004–06'. The plan has the overarching principle to 'ensure that England's soils will be protected and managed to optimise the varied functions that soils perform for society, in keeping with the principles of sustainable development and on the basis of sound evidence' (draft 7 November 2003).

2.23.3 REVIEW OF RELEVANT NATIONAL POLICIES BEYOND EUROPE

2.23.3.1 Australia

The Australian Landcare system offers a possible model for group participation in soil conservation. Landcare began in Victoria in 1986 and has grown to encompass 25 % of the farming community (Campbell, 1995). Landcare adopts an integrated and holistic approach to resource sustainability and is a cooperative venture between Federal and State governments, extension services, consultants and farmers (Curtis and DeLacy, 1995). Over 2700 Landcare groups cooperate on local land degradation issues, which are usually managed at the catchment scale (Hannam, 2000). Issues addressed include identification of land degradation problems and implementation of solutions, development of demonstration sites and promotion of community and stewardship (Ewing, 2000). Education is especially important, in particular, encouragement of 'land literacy' among participants (Campbell, 1995). The scheme has also been successfully adopted in New Zealand (Ministry for the Environment of New Zealand, 1996; Bettjeman, 2000). Some have added cautionary notes over the adoption of Landcare schemes, including the need to maintain fundamental research capabilities (Hannam, 2000) and avoidance of over-optimism on the potential success of the scheme (Bradsen, 2000).

2.23.3.2 North America

In North America, soil erosion research and conservation has been established since the 1930s. A great deal of experience has been accumulated and, consequently, many North American policies, lessons and perspectives have direct relevance to the formulation of European soil conservation policies (Boardman, 1991). In the European context, it should be noted, US and Canadian systems are well structured to deal with erosion policies at national, regional and local levels. Policies are in place, which are largely advisory, but incorporate elements of coercion.

North American soil conservation was galvanised by the severe erosion of the 'Dust Bowl' of the US Great Plains States during the 'dirty thirties' (Bagley, 1979; Hurt 1981). The US Soil Erosion Service, founded in 1933, renamed the Soil Conservation Service (SCS) in 1935, is a permanent branch of the US Department of Agriculture (USDA) (Soil Conservation Society of America, 1980). The US SCS [renamed the Natural Resources Conservation Service (NRCS) in 1994] remains the principal authority responsible for soil

conservation, with over 13 000 employees, and is directly under the jurisdiction of the Secretary for Agriculture. The Soil Survey Division of NRCS provides local and regional soil maps, essential for the formulation of soil conservation strategies. At the NRCS National Soil Survey Laboratory in Lincoln, NE, the Soil Survey Division also analyses samples collected by the staff of around 1000 scientists, thus ensuring accurate and comparable data.

To administer NRCS policies, soil conservation districts are established, usually adopting County boundaries (the administrative subunit of the State). Each district has a conservationist, who is responsible for providing local soil conservation advice. Every district also possesses a 'Board of Supervisors', consisting of unpaid citizens who share an interest in soil conservation. There are 2950 conservation districts, covering most of the 171×10^6 ha of US cropland (Steiner, 1990). Each State appoints a State conservationist, responsible to the NRCS headquarters in Washington, DC.

The NRCS adopts a voluntary land-use planning approach to soil conservation, termed the 'information, education and subsidy' approach (Napier and Napier, 2000a). Upon request by farmers, a conservationist will recommend appropriate strategies for reducing soil erosion to acceptable levels. The NRCS does not possess mandatory powers of land control, but non-implementation of conservation policies can exclude farmers from Federal and State grant-aided programmes. The Conservation Title of the Food Security Act (1985) and the Food, Agriculture, Conservation and Trade Act (1990) introduced elements of coercion into soil conservation policies (Napier, 1990; Esseks and Kraft, 1991). 'Sodbuster provisions' deny farmers access to farm programme benefits if they crop 'highly erodible land' (HEL) without an approved soil conservation plan. 'Swampbuster provisions' aim to prevent conversion of wetland to crop production, while the 'Conservation Reserve Program' (CRP) attempts to retire highly erodible land from agricultural production. The 'Conservation Compliance Provisions' have the greatest potential for long-term reductions in erosion. According to legislation, operators of HEL had to develop an officially approved plan by 1 January 1990, with full implementation by 1 January 1995. Non-compliance resulted in the loss of all USDA farm benefits until an approved plan was implemented (Napier, 1990). The Federal Agricultural Improvement and Reform Act (1996), or FAIRA, increased the emphasis on conservation in current food security legislation (Napier and Napier, 2000a; Weber and Margheim, 2000).

Much can be learnt from North American strategies and policies, but they would need modifying and adapting to European conditions. For instance, set-aside policies were borrowed from North America and established to reduce European grain surpluses (Marsh, 1987; Clarke, 1992; Jones, 1992). However, it is feasible that steep to moderate slopes with erodible soils and other vulnerable parts of fields (i.e. depressions, minor dry valleys and land adjacent to water courses) be put to set-aside (Boardman, 1988; Fullen, 1991a; Chambers and Garwood, 2000). This policy was successfully adopted in the 'Permanent Cover Program' in the Canadian Prairies (Vaisey *et al*., 2000). Preliminary evidence from both the South Downs of England (R Evans and J Boardman, personal communication, 1996) and Shropshire, England (Fullen, 1998) suggests temporary set-aside and permanent grassland are very effective for soil and water conservation. A 'more directed policy' has been proposed in the UK by the Royal Commission on Environmental Protection, which recommended 'the government make maximum use of national discretion to adopt environmental and soil protection criteria in the selection of land for set-aside, and encourage this approach to set-aside at EU level' (Royal Commission, 1996; Recommendation 24).

The impact of set-aside on soil erosion and its off-site consequences is not straightforward, as demonstrated by a modelling exercise for cultivated catchments in central Belgium (Verstraeten *et al*., 2002). For regions with different soil, land-use and morphological characteristics, the impact of the same set-aside percentage results in different erosion rates. Furthermore, decreases in off-site sediment delivery may be different from decreases in soil loss, depending on the location and environment of set-aside fields. Sites prone to soil erosion are not necessarily those fields delivering most sediment to river channels. The optimal location of set-aside fields will therefore depend on policy goals.

In a 50-year review of US soil conservation experience, Sanders (1990) argued that policies must be developed, based on thorough analysis and understanding of the problems, and they should remedy causes of erosion, rather than simply treating symptoms.

2.23.4 SYNTHESIS OF NATIONAL POLICIES: TOWARDS A SOIL CONSERVATION POLICY FOR EUROPE

Successes and failures of various policies developed in the industrialised world (Australia, Europe and North America) were debated at an international conference in Prague in 1996. Based on these discussions, Napier *et al.* (2000a) presented 38 chapters, which thoroughly explore these issues.

Voluntary and non-government organisations have an important role in developing soil conservation policy (Schnepf, 2000). Development of informed debate is critical to future development of European soil conservation and several organisational developments are assisting. The European Society for Soil Conservation (ESSC) was founded in November 1989, with the mission of developing an integrated European approach to issues of soil erosion and conservation. The ESSC consists of a group of scientists attempting to influence governmental policies and public attitudes towards erosion problems (http://www.essc.sk). It consists of 534 members from 46 countries, including 36 European countries (Valencia Congress, March 2000). The initial ESSC Conference, in Coventry in 1990, resulted in a valuable overview of European research (Boardman *et al*., 1990). The first ESSC Congress, held at Silsoe College, Bedfordshire, UK, resulted in a further review (Rickson, 1994). The second ESSC Congress, held at Munich, in September 1996, provided selected papers in a special issue of *Soil and Tillage Research* (1998, Volume 46, Issues 1 and 2) (Auerswald and Kutilek, 1998). The third ESSC Congress, held in Valencia, Spain, in March 2000, focused on highlighting ESSC work for public attention. It offered a timely review of the status of soil erosion and conservation research at the dawn of the new millennium, with the publication of two comprehensive Congress volumes (including 196 chapters), mainly dealing with the European dimension (Rubio *et al*., 2002). The fourth ESSC Congress was held in Budapest in May 2004.

An e-mail based ‘soil erosion discussion group’, coordinated by Purdue University, West Lafayette, IN, USA (e-mail address: se-list@ecn.purdue.edu), provides a valuable means of discussion and communication, linking over 360 subscribers world-wide (Bernsdorf and Favis-Mortlock, 1995).

Increased activity of the International Erosion Control Association (IECA) in Europe is another welcome trend. The IECA was founded in 1972 and based at Steamboat Springs, CO, USA (http://www.ieca.org/). It operates mainly in the USA, but its first European Conference was held in Barcelona in May 1996. The practical outlook of the IECA, with strong emphasis on technical, engineering and industrial solutions to erosion and sediment control, should complement and enhance the activities of more academically orientated European researchers.

Rational policies must be designed at international, national, regional and local scales. At the European scale, the European Soil Bureau (ESB) is a useful forum for discussion of Europe’s soil management problems. This was initiated by meetings of the heads of Europe’s soil survey organisations in 1989 and led to the establishment of the ESB in 1996, based at the EU Joint Research Centre in Ispra, Italy. The ESB focuses on harmonising soil survey operations, but could play an increasingly important role in promoting sustainable development of Europe’s soil resources (http://esb.aris.sai.jrc.it/).

The EC is playing an increasingly active role in developing soil conservation strategies. The 6th Environmental Action Programme (EAP) of the European Union was adopted in 2002 and identified soil as a natural system and a non-renewable resource and established the general objective to protect soil, particularly against erosion and pollution.

The Environment Directorate General of the EC has drafted a policy that 'contains the commitment by the Commission to develop a thematic strategy for soil with the ultimate goal of raising the political importance of soil issues at EU level, so soil protection receives adequate attention'. The 'Soil Protection Communication' (26 October 2001) passed public consultation enabling the policy statement 'Towards a Thematic Strategy for Soil Protection' to be published on 16 April 2002 (Commission of the European Communities, 2002).

The EC is developing a broad overview of soil problems in Europe. It considers erosion to be a critical problem for Europe's soils and believes that soil conservation is a cornerstone of soil protection policy. The Commission recognises that problems in southern Europe are well documented and are clearly evident in the presence of rills and gullies. However, the situation is more insidious in northern Europe. For instance, high erosion rates can occur by sheet erosion, which is usually not considered a significant problem by farmers. Furthermore, associated pollution of waterways by erosion-derived agrochemicals is largely 'invisible' and does not receive sufficient attention. The Commission takes the view that considerable data exist on the nature, extent and severity of erosion on European agricultural soils. The need for more information and monitoring of soil erosion processes, such as gully erosion, piping and soil losses associated with root and tuber harvesting and land levelling, have been identified. The need both to consider the extent and severity of soil degradation problems and to support attempts to develop suitable and innovative soil conservation measures is stressed, including the potential of conservation tillage as a soil conservation measure. There is significant concern about possible implications of the former 'Accession States' joining the EU, as many have considerable soil management problems, particularly associated with chemical contamination (Commission of the European Communities, 2002).

The EC believes that education is important to increase public awareness of soil as a resource and proposes a glossary of soil terms would be useful. The importance of soil as a store for Europe's archaeological heritage is also acknowledged.

The EC identifies areas where there is a paucity of information on Europe's soils. Relatively little is known about long-term trends in soil organic matter contents, the carbon sequestration potential of soil and the capacity of soils to safely absorb sludges. Better understanding is required of linkages between erosion-derived pollution of waterways and threats to the integrity of aquatic ecosystems. There is debate on the potential effects of ecological–organic agriculture on soils, but it is generally agreed it will increase soil biodiversity. The Commission appeals for more data on soil sealing and sterilisation by construction and industrial activities. The considerable heterogeneity of Europe's soils poses problems in developing universally applicable policies. Further challenges are posed by European political diversity, with 156 political units (e.g. Lande, States, Cantons) responsible for policy implementation within the former 15 EU states.

It is recognised that although there are some 41 international conventions dealing with the environment, only one deals with land and is specific to arid environments (UN Convention to Combat Desertification, UNCCD). The Commission is investigating national policies on soil protection (for instance, in Belgium, Denmark, France, Germany, The Netherlands, Sweden and the UK) and attempting to abstract different policy options applicable to Europe. The 'Thematic Strategy for Soil Protection' will be updated regularly, in consultation with interested parties, including the 'Stakeholders Forum' established in January 2002. This strategy is a permanent and integral component of the Commission's overall environmental and agricultural policy. The text is available at the EC web site: http://europa.eu.int/comm/environment/soil/index.htm

Interested members of the public can express their views on soil policy to the EC via the web-based 'CIRCA-Soil' (the soil policy electronic forum), launched in July 2003, at http://forum.europa.eu.int/Public/irc/env/Home/main

In spring 2003, the EC established five technical working groups with the main goals of gathering existing information on the eight identified soil threats and making recommendations for measures and policy options.

Three soil priority areas were defined (erosion, contamination and organic matter), constituting the core issues of three thematic working groups. Further, two horizontal working groups on monitoring and research were established. The other five soil threats were dealt with by the most related working groups: compaction, floods, landslides and salinisation were linked to erosion, biodiversity was linked to organic matter and sealing by construction was linked to research. The final aim of the working groups is to contribute to EC deliverables. However, what will be delivered depends on political decisions taken by the new Commissioner, appointed in November 2004. It will be a single package, including a communication on actions, and possibly a Soil Framework Directive. The EC will also formulate directions for future research.

Generally, especially from within the Working Group on Soil Erosion, the development and implementation of a Soil Framework Directive was supported. This was in order to reach a more coherent and visible soil policy, rather than only linking up soil issues with other different policies. This would imply legally binding measures to protect soils from threats and the obligation for Member States to monitor the state and trends of soils with respect to these threats. The foundation of national and/or a European Soil Conservation Service was considered necessary for the proper implementation of such a Directive. The amendment of existing policies to address soil threats better, e.g. through the Water Framework Directive or the CAP, was also suggested. After the technical working groups finished their reports (15 May 2004), the consultation phase was finished. The future development of the EU Soil policy will then largely be determined at the political level, but at the administrative and expert level the protection of soils is acknowledged as a high-priority concern.

Governments have crucial roles to play, especially at a national level (Napier *et al.*, 2000b). They should direct their policy to monitor the status of their national soils and to achieve proper soil use and conservation. A commendable example of a clearly formulated policy statement is the 'Sustainable Land Management Strategy' of the New Zealand Government (Ministry for the Environment of New Zealand, 1996; Bettjeman, 2000). There is debate whether government actions should be on a cooperative and voluntary basis (e.g. Johnson, 2000), or whether there should be a more coercive and regulatory approach (e.g. Napier and Napier, 2000a,b). Where possible, government involvement should not be authoritarian or punitive, but should aim at facilitating conservation by assisting in the identification of problems, in tackling the underlying causes of soil misuse and by encouraging necessary actions. A senior administrative body or commission would be necessary for such a task. The body should, in consultation with interested parties, establish, promote and finance research priorities. However, if all reasonable attempts at cooperation fail, then a more coercive approach would be necessary. Current emphasis on the 'polluter pays' principle goes some way towards addressing this issue. The responsible body should have clear and verifiable aims. For instance, the 'Ontario Land Stewardship Program' (OLSP) in Canada has been criticised for lacking specific and measurable aims (Stonehouse, 2000).

Availability of accurate, high-quality soil data is pivotal to a successful policy. A European and national inventory of land resources is necessary, so gaps in knowledge are identified and, where necessary, studies commissioned. National soil survey organisations must play a vital role in providing information (Young, 1991). Bullock and Thompson (1996) argued for a two-stage integrated policy for improving the sustainability of UK soil resources. First is the identification of the current state of soils, to assess the capability, vulnerability, sensitivity and resilience of soils, to inform the decision-making process. Second is to match soils and their use, so land use is sustainable and appropriate.

European governments have tended to regard national soil survey organisations as rather esoteric entities, and their finances are often vulnerable to the whims of finance ministries. Much can be learnt from the US experience, where the Soil Survey is a respected, properly funded organisation, with a relatively high profile in public awareness (Batie, 1985). European Soil Surveys should consider adopting the policy of US County soil surveys of incorporating USLE soil erodibility (K) value assessments into their mapping at the series scale. This approach has proved useful for regional erodibility mapping in Belgium (Pauwels

et al., 1980), Denmark (Madsen *et al.*, 1986) and Germany (Becher *et al*., 1980; Jäger, 1994). In addition, temporal trends in organic content of arable soils should be closely monitored (Fullen, 1991b). The NSRI evaluated changes in soil organic matter between the early 1980s and 1995 and found that concentrations are decreasing on arable topsoils and many former peats are now classified as humose mineral soils (Bullock and Burton, 1996).

Paucity of information on the costs of erosion impedes full evaluation of its effects. Data collection and evaluation of both 'on-site' and 'off-site' costs are problematic. Many costs are difficult to quantify, borne by various groups (e.g. local councils, water authorities, insurance companies and householders), are inherently difficult to collate and are not necessarily costed as being directly due to soil erosion (Stammers and Boardman, 1984). Costing of erosion and related flooding episodes on the South Downs of England placed 'off-site' costs between €90 000 and €350 000 and up to almost €1.8 million at Rottingdean in 1987 (Boardman *et al*., 1994). A tentative costing of both 'on-site' and 'off-site' erosion in England and Wales produced a combined total of €1,400 million per year (Evans, 1995). The 'off-site' cost associated with downstream sedimentation is considered to be the dominant problem (ADAS, 2002; Boardman, 2002). Similar 'off-site', high-cost scenarios of erosion episodes have been suggested in central Belgium, including non-quantifiable social costs of stress, induced by flood damage and risk to property (Verstraeten and Poesen, 1999). Comprehensive and accurate costing of soil erosion would be helpful, both in evaluating the problem and in planning policy responses. Adoption of the 'polluter pays' principle would promote more effective conservation.

On a regional scale, skilled personnel are necessary for consultative duties. With reference to the UK, Boardman (1988, 1991) suggested the establishment of a small soil conservation unit within MAFF. Morgan and Rickson (1990) argued the Danish Land Development Service (Hedeselskabet) or the Landgraedsla Rikisins of Iceland could act as organisational models generally within Europe. Establishment of European soil conservation services, whether as distinct entities or as subdivisions of agricultural advisory services, merits discussion. Essential components of any soil conservation service should supply free information and advice to agriculturalists and interested bodies. Advisory services should also freely disseminate information to the public, particularly educational establishments. Current UK government policy of charging fees for advice from DEFR is counterproductive. It is imperative that soil conservation field demonstrations are organised, so farmers can see tangible evidence of conservation benefits. As erosion occurs on a field scale, local conservation policies are essential (Evans, 1990). Agriculturalists must be able to call freely upon the advice and expertise of soil scientists. In this respect, Europeans have much to learn from the US NRCS. Advisors should assist farmers, identifying causes of erosion and selecting appropriate technologies for remediation. Evolution of a conservation plan should be an interactive process between an advisor and agriculturalist, leading to the development of a range of possible costed strategies. Costs of remedial or preventative measures may be prohibitive, therefore the US cost-share system, where the NRCS can meet up to 75 % of the cost of conservation measures, seems appropriate. A similar approach is adopted by the Icelandic SCS (Arnalds, 1999), the Canadian OLSP (Stonehouse, 2000) and by the Ministry of Flanders, Belgium (Vandekerckhove *et al*., 2003). With respect to US agriculture, it was argued that 'environmental credits' could be introduced to recognise and financially reward actions of producers who voluntarily introduce conservation systems on their land (Weber and Margheim, 2000). Sanders (1990) stressed the need for local voluntary organisations to discuss erosion problems. The US County soil conservation district could provide a useful model. In such a forum, interested parties meet and discuss local erosion problems and potential solutions.

Education plays an essential role in informing the public on the importance of soil as a resource. Several illustrative examples of good practice can be identified. These include the US NRCS policy to identify 'State Soils', that is, a specific soil type associated with each State. These are used in school education programmes, with students visiting representative profiles. Soil education was a major component of the EXPO 2000

Exhibition, near Hannover, Germany, held from June to October 2000 (Auerswald, 2000; http://www.obe2000.de). In Devon, England, the Institute of Grassland and Environmental Research (IGER) has made important contributions, including the production of an educational resource leaflet for teachers and students entitled 'Working with Soil'. IGER has also developed educational 'Soil Trails', designed on the basis of 'Nature Trails'.

It is imperative that the broader benefits for soil conservation to society are recognised. Not only is conservation beneficial to agricultural development, it also assists several environmental objectives. For instance, soil conservation is compatible with habitat creation and the promotion of biodiversity. Soil conservation can assist with the accumulation of soil organic matter, which is an important 'sink' for atmospheric carbon. In turn, carbon sequestration can help ameliorate global warming (Wedin and Tilman, 1996; Arnalds, 1999; Lal, 2000).

2.23.5 CONCLUSIONS

The extent and severity of erosion on European soils have increased markedly over the last 50 years, particularly on arable land. However, government action and advice on soil conservation have been limited. Recently, government and agency interest in soil erosion has increased, largely owing to 'off-site' costs and problems associated with downstream sedimentation and water quality. Taking countries representative of different regions of Europe, a broad range of instruments are used, usually at the catchment scale. These include agri-environment measures, legal instruments, erosion control plans, the promotion of participatory approaches, subsidies, education programmes and the development of advisory services. These instruments are promoted to varying degrees, in response to both the severity of erosion problems and the degree of government interest and involvement. In this review of strategies to improve soil conservation, several 'best management' practices are identified. Policies at international, national, regional and local scales should include:

1. Initiation of national soil conservation services. These organisations should be properly-funded and relatively well publicised.
2. Full mapping, monitoring and costing of erosion risk by national soil survey organisations.
3. A participatory approach to soil conservation, involving farmers and interested members of the public.
4. A 'cost-share' partnership between government and farmers.
5. Development of rational land-use policies, such as targeting temporary and permanent set-aside on steep and erodible land, use of grass strips on erodible arable slopes and the protection and management of riparian zones.
6. Increased public understanding and awareness of the value of soil resources through education programmes. Education schemes should particularly encourage 'land literacy' among participants.
7. Broader benefits of effective soil conservation to society should be recognised and promoted, such as its potential contribution to habitat creation, promotion of biodiversity and carbon sequestration.

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2.24

Agri-environment Measures and Soil Erosion in Europe

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2.24.1 INTRODUCTION

All present pan-European soil erosion assessment methods and accompanying maps agree that the highest erosion rates occur in southern Spain, Italy, Sicily, Sardinia and Greece (see Chapter 2.13). An analysis of the PESERA map per country shows that 16.7 % of the EU-15, excluding Sweden and Finland, is susceptible to considerable erosion risk with highest rates occurring in the Mediterranean area (Figure 2.24.1). In comparison, the EEA estimated that about 12 % of the EU-15 (115×10^6 ha) is prone to risk of water erosion and 4 % of the EU-15 (42 million hectares) is prone to wind erosion (EEA, 2001). Furthermore, the highest erosion rates are observed and predicted by erosion models on agricultural land and particularly on arable land (Gobin *et al.*, 2003). With about 62 % of agricultural land under intensive agricultural cultivation (IEEP, 1995; Bignall and McCracken, 1996, 1995), many European farmers' fields are at risk of serious soil degradation.

In Mediterranean Europe, the problem of maintaining, or improving, the physical, chemical and biological quality of soil, while reducing soil erosion, is particularly difficult to solve. The strong risk of soil loss is caused by the high rainfall intensity and the frequent occurrence of extreme events, by the high relief energy and the vulnerable condition of soil when erosive events occur. Furthermore, the intense agricultural exploitation of soil on highly mechanised farms, drought conditions, forest fires, land abandonment in marginal areas and overgrazing increase the risk of soil degradation (Wells, 1981; Albaladejo *et al*., 1991;

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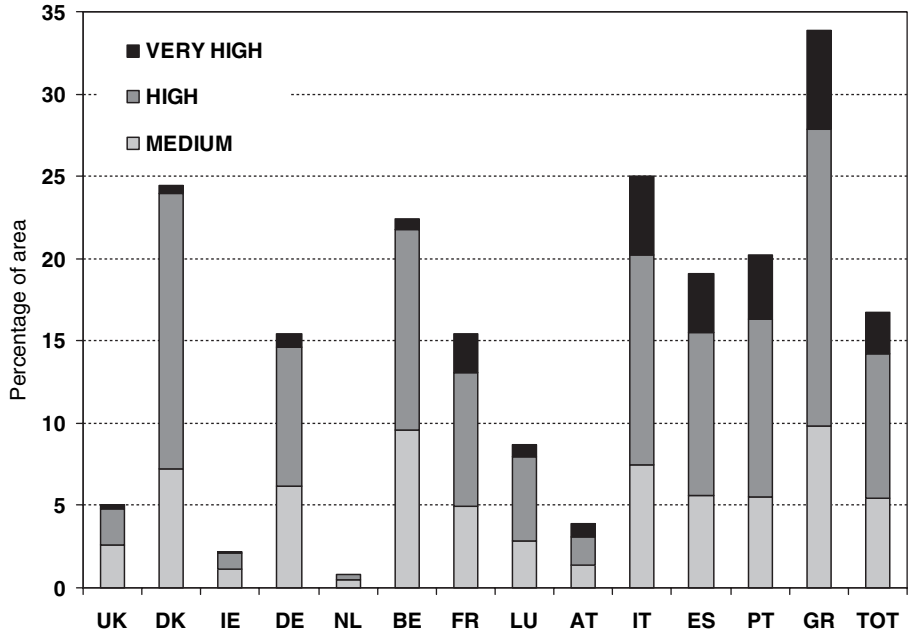


Figure 2.24.1 Risk of erosion by water in Europe, based on PESERA predictions

Kosmas *et al.*, 1996). High air temperature, combined with ploughing and harrowing, favours the loss by mineralisation of soil organic matter (Kirschbaum, 1995; Kätterer *et al.*, 1998), which is a key factor for conserving soil health (Kononova, 1966; Follett, 1987; IPCC, 2000).

Certain trends or changes in European agriculture such as technological improvements, agricultural land use and livestock management greatly influence soil erosion. Technological and transport improvements have encouraged agricultural specialisation in areas with the lowest production costs, with consequent intensification of chemical and mechanical inputs. The reforms that have been introduced by the European Common Agricultural Policy (CAP) have led to land-use changes that may affect land degradation and in particular erosion. Agricultural production has been artificially constrained by mechanisms such as milk quotas and compulsory set-aside for arable crops. Although this is *prima facie* environmentally beneficial, many of the statistics show otherwise.

Over the past 20 years, the area of land under productive agriculture has fallen by 2.5 %. An analysis between Farm Structure Survey (FSS) Census data between 1990 and 2000 shows that all-agricultural land-use classes decreased in area, but the amount of arable land increased at the expense of permanent grassland. The class permanent grassland declined by 5 %, permanent crops by 4 % and arable land only by 0.7 %. Traditional crops in rotation declined and mono-crop farming systems increased. Particularly fodder maize for silage increased, with a consequent increased risk of soil erosion.

Oldemann *et al.* (1991) demonstrated that overgrazing is one of the major causes of soil degradation worldwide. Based on FSS census data for 1990 and 2000, sheep stocks declined by 3 % and cattle by 5 % throughout the EU-15. However, these figures hide national differences. The countries where overgrazing is most risky in terms of soil degradation (soil erosion, compaction and landslides) are located in Mediterranean Europe. In 1995, nearly 50 % of goats were located in Greece, with another 33 % in Spain and Italy. About 40 % of EU sheep are in Greece, Spain and Italy (EUROSTAT, 2001).

A policy of subsidies for fallow land (set-aside) was introduced to contribute to market balance by reducing surplus production. Crops intended for non-food use (biomass and biofuels) are permitted on set-aside land. In cases where set-aside land is used for non-food row crops (industrial set-aside) or when the land remains tilled and uncultivated, soil quality may decrease and the risk of soil erosion may increase. In contrast, when set-aside land remains covered by natural or sowed grass, the physical and chemical soil properties can ameliorate. Between 1975 and 1997, the area under cereals decreased by 7 % in the EU-9 as a result of set-aside. However, industrial set-aside increased from 11.2 % in 1999 to 15.5 % of the total set-aside in 2003. The total area of set-aside land, whether rotational or non-rotational (fixed) and voluntary or not, increased by 34 % between 1997 and 2003.

2.24.2 AGRI-ENVIRONMENT PROGRAMMES AND MEASURES

Agri-environment programmes were introduced into the CAP during the mid-1980s as an optional policy instrument to support specific farm practices that help to protect the environment and maintain the countryside. Agri-environment programmes aimed to cover all aspects of agricultural activity that interact with the environment, i.e. air, soil, biodiversity, landscape, land and water. In particular, they focused on environmental problems on farmland under intensive agricultural systems and the management of adjacent zones, such as field margins.

The McSharry reform of the CAP in 1992 introduced agri-environment programmes throughout the territory (Council Regulation No. 2078/92). These programmes stimulate and pay farmers, for a 5-year minimum period, to produce environmental services through environmentally beneficial activities (EEC, 1998) that go beyond 'good farming practice'. In response to Regulation 2078/92, the European Commission approved more than 130 different programmes containing over 2000 distinctive measures presented by the 15 Member States prior to 1998.

In 1999, the Agenda 2000 reform included agri-environment measures as an obligatory part of rural development programmes (Council Regulation 1257/1999) to be designed at national, regional or local level. Agri-environment measures are the main instrument for the integration of environmental concerns into the CAP. Examples of commitments covered by national/regional agri-environment measures include environmentally favourable extensification of farming, management of low-intensity pasture systems, integrated farm management and organic agriculture, preservation of landscape and historical features such as hedgerows, ditches and woods and conservation of high-value habitats and their associated biodiversity.

European agricultural regions vary widely with regard to landscape, farming types/structures and agri-environment issues, resulting in corresponding differences in specific regional requirements. This in turn leads to differences in implementation; it is therefore clear that the evaluation of agri-environment measures and programmes depends on the development and application of methodologies which are both regionally specific and, only to a certain extent, cross-nationally comparable. An additional difficulty is that there are very few explicit data on what is really happening, more in particular on the links between certain agricultural practices and soil erosion.

In 1998, 20 % of the utilised agricultural area in Europe was covered by agri-environment measures (Figure 2.24.2); in 2002, this was 25 %. The target in the 5th Environmental Action Program (EAP) of at least 15 % of EU farmland under agri-environment agreement by 2000 was exceeded, but implementation remained below 15 % in six Member States in 1998 (EEC, 1998). The 2002 data clearly show the upward trend in the uptake of agri-environment measures, apart from Italy and Germany, where a downward trend is observed. Luxembourg, Austria, Finland and Sweden clearly outperform the European average, whereas Greece, The Netherlands and Spain remain far below the European average.

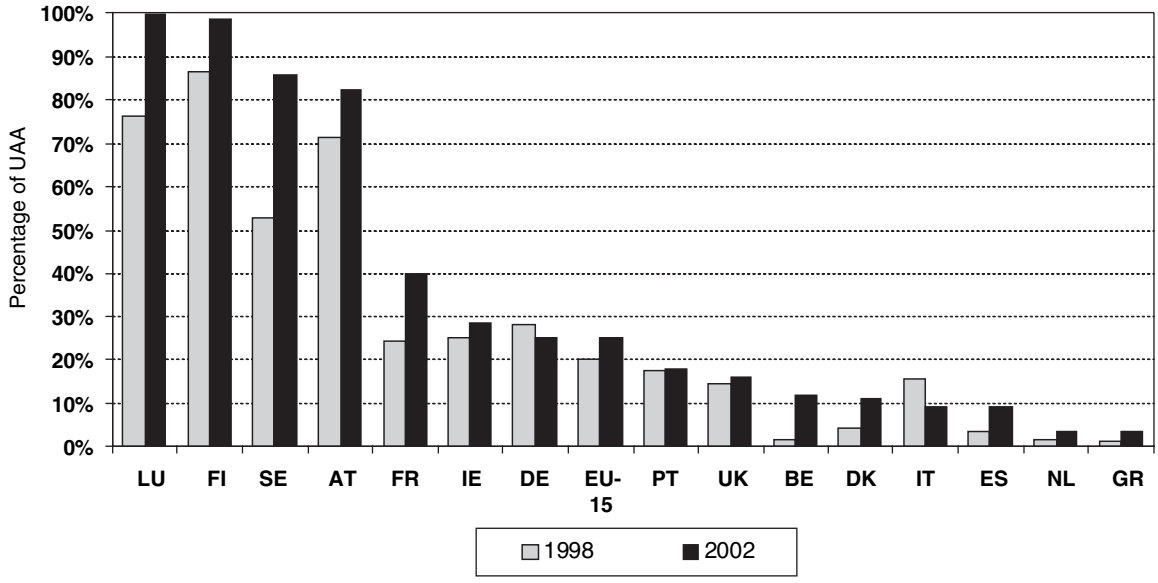


Figure 2.24.2 Percentage of utilised agricultural area (UAA) under agri-environment measures (AEM) in 1998 and 2002

The different agri-environment measures are commonly divided into the following groups: organic farming; input reduction, crop rotation and extensification, landscape and nature; and plants under genetic erosion and other more site-specific measures. For the EU-15, input reduction, crop rotation and extensification account for 32 % and organic farming for 6.5 % of the total area under agri-environment measures (Figure 2.24.3). Landscape and nature (15 %) and other measures (40.5 %) are region specific and are therefore more difficult to discuss in a European-wide context. Plants under genetic erosion (6 %) are obviously not related to soil erosion. Most of the Member States have agri-environment measures related to organic farming, input reduction and extensification.

2.24.3 AGRI-ENVIRONMENT MEASURES AFFECTING SOIL QUALITY

2.24.3.1 Organic and Integrated Farming Systems

Organic farming is an approach to agriculture where the aim is to create integrated and environmentally and economically sustainable agriculture systems (Lampkin and Padel, 1994; Rigby and Cáceres, 2001). Integrated farming systems put together technologies to produce site-specific management systems for whole farms, incorporating a higher input of management and information for planning, setting targets and monitoring progress (Pretty, 1998). Integrated farming represents a step from high-input farming to organic agriculture.

The aim of the organic farming measure, first introduced and regulated by EEC Regulation 2092/91, is to help farmers manage ecological and biological processes in a framework of self-regulating agro-ecosystems that use locally or farm-derived renewable resources. The use of external inputs, whether inorganic and organic, is reduced as far as possible. Organic farms have lower inputs and concentration of nutrients compared with conventional farming practices, which leads to reduced leaching into water and reduced emissions to the atmosphere. The increased input of organic matter, broad rotations and omission of pesticides contribute to the protection and preservation of soil biodiversity (Mader *et al.,* 2002). Indicators of soil quality,

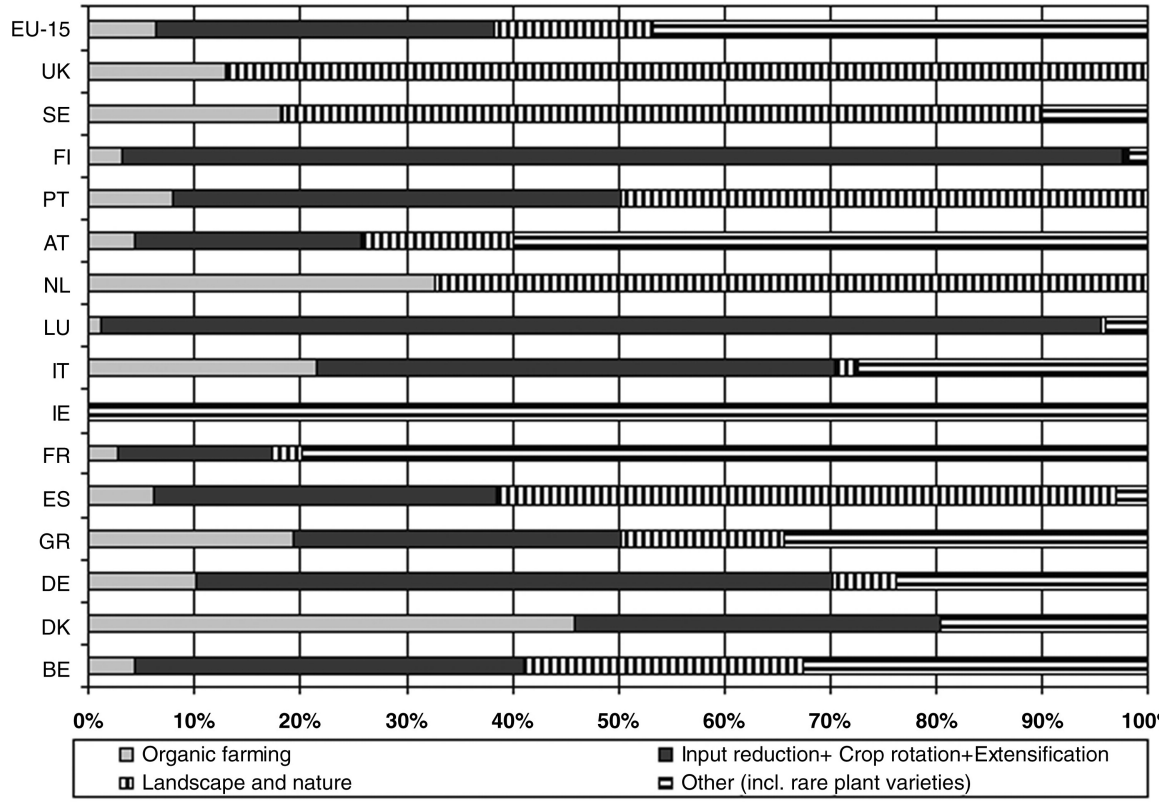


Figure 2.24.3 Division according to type of agri-environment measure (2002)

e.g. dehydrogenase activity (DHA) and bacterial and fungal counts, are higher on soils under organic management than on inorganically fertilised soils (Bardgett *et al.,* 1997). Long-term benefits associated with organic farming include greater topsoil depth, more moisture retention and reduced soil erosion (Reganold *et al.,* 1987).

The environmental impact of 29 agri-environment measures was compared against the reference standard of good agricultural practice (study by the University of Göttingen for DG-Agriculture; EEC, 1998) in terms of preservation of water, soil, atmosphere, diversity of fauna, flora, biotopes, landscape and cultural aspects. The results showed that measures with a high positive impact on the preservation of environmental resources include the omission of fertilisers and plant protection products (pesticides), especially in environmentally sensitive areas, and the conversion of arable land to extensive grassland. The *status quo* of agricultural management performed worse than good agricultural practice and integrated farming slightly ameliorated environmental resources. The impact of integrated farming in low mountain ranges was less significant due to the fact that extensive farming methods are already applied in these areas.

In Tuscany, a study was carried out by Pacini *et al.* (2003) to evaluate the sustainability of organic, integrated and conventional farming systems on three farms with widely ranging climatic and soil conditions. The study applied a holistic, integrated economic–environmental accounting framework through the measurement of a number of indicators and pedo-climatic variables. Organic farming systems performed better than conventional and integrated farming systems and were more profitable. However, the environmental responses of the three different systems are highly affected by pedo-climatic factors both at regional and farm scale.

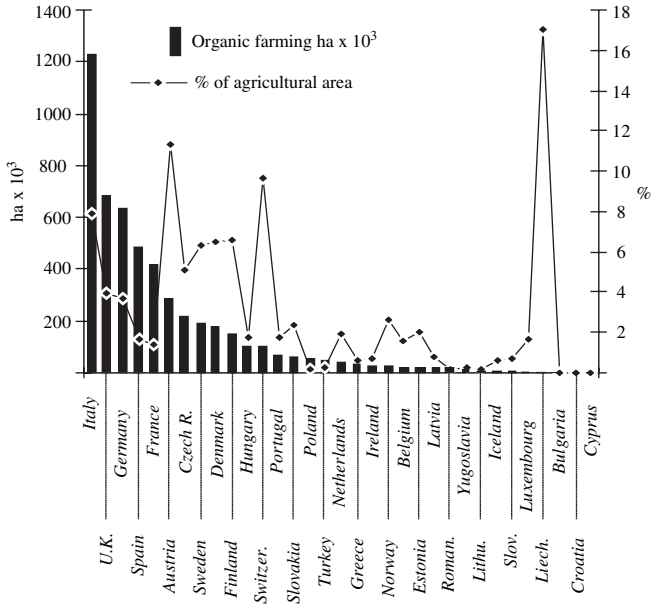


Figure 2.24.4 Hectares and percentage of utilised agricultural area under organic farming in Europe in 2001

Data published by three services of the European Commission: Eurostat, DG-Agriculture and DG-Environment (EU, 2003), show that in 2000 (Figure 2.24.4), the area devoted to organic farming (fully converted and in conversion) covered 3.8×10^6 ha in the EU-15, whereas in 1998 it covered only around 2.3×10^6 ha. This represents an increase of 67 % over the period 1998–2000 and can be considered a reliable indicator of soil quality improvement. The organic farming area reached 3.0 % of the total utilised agricultural area (UAA) of the EU-15 in 2000, up from only 1.8 % in 1998.

The dominant organic crops in the EU's Nordic countries (Denmark, Finland and Sweden) in 2000 were cereals, forage plants, pastures and meadows, each covering around one-quarter of the combined organic area. The prevalence of these crops has fluctuated only slightly since 1998, although the total organic area has increased by almost 50 %.

In western European countries (Austria, Belgium, Luxembourg and The Netherlands), organic farming on pastures and meadows covered around 75 % in 2000, whereas both cereals and forage plants remained below 10 %.

In southern Europe (France, Greece, Italy, Portugal and Spain), the organic area has grown by around 70 % during the period 1998–2000 (Figure 2.24.5). Dominant organic crops in 2000 are pastures and meadows, forage plants, cereals and olive plantations. Vineyards represent 3 % of the organic crops.

In Austria, Finland and Italy, the increase in organic farming area has been due to, or coincided with, expansion of the agri-environment programmes (Figure 2.24.2). The countries with an increase in organic farming area lower than the EU-15 average were The Netherlands, Luxembourg, Spain, Germany, Finland, Sweden and Ireland. Austria saw a small decrease in its organic farming area since 1998, after high growth figures in previous years.

More than one-quarter of the total area devoted to organic production in the EU-15 in 2000 was located in Italy. The UK is the second best organic producer, followed by Germany, Spain and France.

The countries with an increase in organic farming area in the period 1998–2000 above or close to the EU-15 average were the UK, Italy, Belgium, Greece, Denmark, France and Portugal. The high growth figures for the UK are mainly due to recent conversions of very large, very extensive holdings in Scotland.

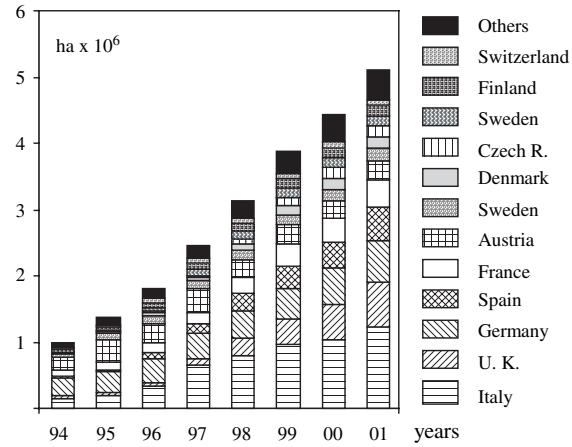


Figure 2.24.5 Trend of organic farming in EU countries during 1994–2001

2.24.3.2 Extensification and Input Reduction

Extensification by converting arable land into grassland is effective for improving soil quality (Doran *et al.*, 1996; Fauci and Dick, 1994). In the EU-15 there are about 45×10^6 ha of grassland. Under Regulation 2078/92, 0.5 % of arable land areas, corresponding to 366 000 ha, received subsides for conversion into pastures (Table 2.24.1).

The Netherlands and Denmark were the main beneficiaries of this measure. In 1997, they alone accounted for 82 % of the areas in the EU converted and assisted under this agri-environment programme. The average area converted into pasture was relatively small in Denmark (5.2 ha) and concerned almost exclusively holdings of more than 10 ha.

In Spain and Portugal, few areas and holdings received this type of aid, but the average area of arable land converted was considerably greater: 49 ha per holding in Spain and 21 ha per holding in Portugal.

The area under specific management contracts that gives more attention to maintaining biodiversity and landscape represents another indicator of environmental development of agricultural systems and of soil conservation. These contracts cover more than 22×10^6 ha (20 % of the utilised agricultural area of the EU). The extent of the contract varies between Member States: from more than 60 % of farms in Austria, Finland and Sweden to 7 % or less in Belgium, Greece, Spain and Italy. However, area alone gives no indication of the environmental performance of the agri-environment programme, as many of the programmes are not assessed in terms of effectiveness. Figure 2.24.6 shows two examples of agri-environment measures adopted in the Chianti area to conserve the soil and landscape.

With respect to the total CAP budget, expenditure on management contracts remains extremely modest (only 4 % of the European Agriculture Guidance and Guarantee Fund). It is sometimes more profitable for farmers to receive an EU payment for arable set-aside than to enter an agri-environmental programme (EEA, 2000).

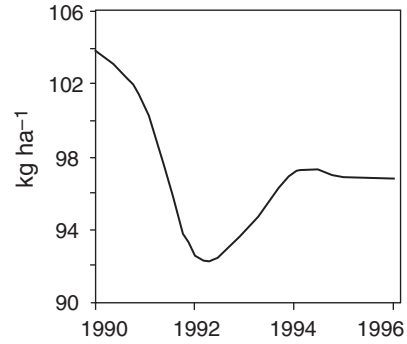
Reduction of fertiliser (Table 1) during the application of Regulation 2078/92 is another indicator of extensification and improvement of soil quality. The use of nitrogen and phosphorus fertilisers has decreased overall, but this trend has been slightly reversed since 1992.

2.24.4 DISCUSSION

An evaluation of Agri-Environment Programmes (Working Document VI/7655/98 of DG Agriculture; EU, 1998) showed that positive effects of agri-environment measures on soil quality have been achieved. These

TABLE 2.24.1 Trend of fertiliser use and area of arable converted to permanent grassland and meadow (EUROSTAT, 2001b)

Country	ha	% ha in farms ≥50 ha (UUA)	Total nitrogen and phosphorus fertilisers (Liechtenstein not included)
Netherlands	15220	30	
Denmark	14740	74	
Spain	28650	97	
France	12630	66	
Ireland	9200	44	
Italy	8260	55	
Portugal	7240	85	
Austria	350	3	
Belgium	300	70	
Luxembourg	150	100	
Greece	10	0	
EU-15	366	56	



beneficial effects are the consequence of the highly positive results reported for reduced inputs, organic farming, nature protection and measures to maintain landscapes.

Some difficulties arose with extensification over the past 20 years. The measure was rarely applied because farmers did not accept long-lasting periods of non- or reduced cultivation.

Measures to convert arable land to grassland, or to ensure mixed farming and rotation, performed better than segetal vegetation (row crops, small grains) in terms of soil conservation. Considerable evidence of environmental benefit resulted from erosion prevention measures, such as mulch seeding.

Nature management often requires low-intensity grazing of pastures with consequent positive impacts on soil protection. However, extensification of livestock has not been very successful in several regions; one reason may be that it is not financially rewarding. Maintenance of extensive systems essentially failed in many important zones of the EU-15.

To evaluate the effectiveness of agri-environment programmes in terms of soil quality, an effort is needed to improve data collection. In fact, existing monitoring programmes do not provide sufficient data to show the trends of physical, chemical and biological properties of soil and the effects on soil erosion, landslides and off-site impacts. Under EC Regulations, each EU-15 Member State collects highly detailed geo-referenced information on land use at the field scale to check tax declarations and compile statistics. From these databases, it is possible to track soil use and management during the years of application of agri-environment measures. Furthermore, these data can give the possibility of performing scenario analysis. However, such information is currently not accessible owing to confidentiality rules.

Figure 2.24.6 Example of agri-environment measures adopted in the Chianti area on vineyards. (A) The existing non-economic stonewall terraces (see B) were enlarged and connected (zig-zag paths) to help mechanisation. The alternate removal of a stonewall row every two terraces maintained the landscape function and reduced the slope length respect to site C, where land levelling was adopted. (C) Grass cover applied to a vineyard planted along the maximum slope, to reduce erosion (*photo* P. Bazzoffi)

The erosion literature commonly identifies acceptable rates of soil erosion that typically range from 1 t ha^{-1} yr^{-1} on shallow sandy soils to 5 t ha^{-1} yr^{-1} on deeper well-developed soils. However, with a very slow rate of soil formation, and considering that tolerability of soil erosion must be viewed with reference to off-site socio-environmental costs and benefits, any soil loss of more than 1 t yr^{-1} can be considered to cause irreversible damage within a time span of 50–100 years (EEA, 1998). Although soil is a vital and largely non-renewable resource, it has not been the subject of comprehensive EU action so far. A thematic strategy for soil protection, which recognises soil erosion as one of the major threats, has currently been placed high on Europe's political agenda.

Assessing and monitoring soil erosion are needed to evaluate the impact of, *inter alia*, agricultural and land use policies such as the agri-environment measures in Europe (Gobin *et al.*, 2004). Europe's agri-environment policies are increasingly focusing on gains in environmental functions, by decreasing negative and increasing positive externalities from agriculture (Regulation EU 1782/2003). These externalities, which include soil erosion, need to be quantified and compared against environmental standards of sustainability (Regulations EEC No. 2078/1992 and EEC No. 1782/03). In 2004, as part of the environmental standards of sustainability (Regulation EEC No. 1782/03), the Italian Ministry of Agriculture tried to solve the problem of identifying where soil erosion occurred above tolerable limits. Erosion is considered tolerable on condition that rills do not appear on the field surface. This indicator addresses the need of providing a sure and certifiable reference point to the 'on-site' controls randomly made by the paying organisms. This definition of tolerable soil erosion represents a valid compromise, to be considered by other EU countries, because it is almost impossible to measure soil erosion that occurred in the absence of control devices previously installed in the field.

Since erosion is patchy in both time and space, there is a need for a revised and comprehensive framework to analyse the complexity of the problem. Gobin *et al*. (2003, 2004) proposed a revised driving force–pressure–state–impact–response (DPSIR) framework and a set of soil erosion indicators that can be objectively calculated, validated against measurements or observations and evaluated by experts. The soil erosion indicators have to be evaluated against the physical background of topography, climate, soil characteristics and vegetation cover.

2.24.5 CONCLUSIONS

At the beginning of the CAP, the major issue was to support product prices through compensatory payments, with agri-environment programmes receiving only a relatively small share of the budget. This policy determined the intensification of inputs with potential harmful effects on soil conservation. Various reforms of the CAP have aimed at resolving some of the environmental problems introduced by agriculture.

Quantitative values of change in physical and chemical properties of European soils under the effects of agri-environment measures are lacking. Nevertheless, it is possible to provide a judgment in terms of extent of utilised agricultural area (UAA) where certain agri-environment measures have been applied that influence soil protection. Percentage uptake of organic farming, integrated farming, input reduction, extensification and crop rotation are specific agri-environment measures that reduce soil erosion and, at the same time, are cross-nationally comparable. Other agri-environment measures such as those related to preservation of nature and landscape have a positive influence on soil protection but are site-specific and therefore more difficult to compare across Member States.

Since many of the agri-environment measures have a positive impact on soil protection and combating soil erosion, there is an urgent need for a scheme to monitor soil erosion under different agro-ecosystems in order to evaluate numerically the effectiveness of agri-environment measures. This requires a revised DPSIR framework, a set of soil erosion indicators and measurements/observations for full analysis of the soil erosion problem in different agro-ecological zones and agri-environments (Gobin *et al.*, 2004).

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